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Effects of planted cover management on buffer prey and their role in waterfowl nest success

by

Barbara Anne Maile

A thesis submitted to the Faculty of Graduate Studies and Research in partial fulfillment of the

requirements for the degree of Master of Science

in

Conservation Biology

Department of Renewable Resources

Edmonton, Alberta

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September 26, 2003

University of Alberta

Faculty of Graduate Studies and Research

The undersigned certify that they have read, and recommend to the Faculty of Graduate Studies and Research for acceptance, a thesis entitle Effects of Planted Cover Management on Buffer Prey and Their Role in Waterfowl Nest Success submitted by Barbara Anne Maile in partial fulfillment of the requirements for the degree Master of Science.in Conservation Biology.

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25 Sept, 2003

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ABSTRACT

Recent focus has been directed at the potential for alternative prey species to buffer waterfowl nest depredation since many nest predators also select small mammals and invertebrates. We studied relationships between management of planted cover, vegetation, landscape attributes, nest survival, and small mammal and beetle abundance and biomass. Forty-five projects managed as waterfowl nesting habitat were sampled in the Aspen Parkland of Saskatchewan and Manitoba in 2000 and 2001. Models that included vegetation characteristics but excluded management effects were the most parsimonious for vole, shrew, and total small mammal abundance, based on Akaike's Information Criterion (AIC_c). Small mammal abundance increased with vegetation density for all mammal groups. Beetles abundance was explained by duff depth and conserved soil moisture rather than management. Using an extension of the Mayfield method, nest success probabilities were modelled using AIC to select the most parsimonious model. The best models included vegetation height, wetland area, and beetle and vole abundance parameters. We found a positive relationship between vole abundance and nest success and a negative relationship between beetle abundance and nest success.

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LIST OF ABBREVIATIONS

AIC, Akaike's Informaiton Criterion
AICc, Akaike's Information Criterion, adjusted for small sample size
APH, Alternative Prey Hypothesis
AVE, avian predator activity estimate
BEET, total beetle abundance
CARB, Carabid (ground beetle) abundance
BCIO, Carabid (ground beetle) biomass
CE, corrected catch effort
CPUE, catch per unit effort
CROP, percent adjacent cropland
CSM, conserved soil moisture
DNC, dense nesting cover, amount of planted cover in a project
DSR, duck nest daily survival rate
DUC, Ducks Unlimited Canada
DUF, duff depth
FORB, percent forb composition
GEO, geographic location of study site
GPS, Global Positioning System
K, number of estimable parameters included in models
MAM, mammalian predator activity
MICE, deer mouse abundance
MXHT, maximum vegetation height
NAWMP, North American Waterfowl Management Plan
NULL, null model
PPR, Prairie Pothole Region
SAS, Statistical Analysis System
SHRW, shrew abundance
TN, trap night
TRT, treatment
UTM, Universal Transverse Mercator
VOLE, vole abundance
VOR, visual obstruction reading
WA, wetland area
YPM, year post-management
YR, sample year

CHAPTER 1

GENERAL INTRODUCTION

Introduction

Grasslands and aspen parklands constitute the Prairie Pothole Region (PPR) of the northern Great Plains in Canada and the United States. The PPR encompasses 77 million ha extending from Alberta to Minnesota (Batt et al. 1989). Characteristics of the PPR can be found in Greenwood et al. (1995). Dabbling duck (tribe Anatini) production within the region has historically been high with 50 - 80% of production from an area representing 10% of available nesting habitat (Baldassare and Bolen 1994). Fluctuations in waterfowl populations occur naturally but Beauchamp et al. (1996*a*) suggest nest success has decreased since nesting studies began in the 1930's. Of the many limiting factors in prairie duck populations, predation during the breeding season is the most critical (Klett et al. 1988; Sargeant and Raveling 1992). With limited high quality habitat, predators and nesting ducks have been forced to concentrate into small or highly fragmented patches.

Habitat loss on the Canadian prairies has been ameliorated through permanent cover programs implemented under the auspices of the North American Waterfowl Management Plan (NAWMP; U.S. Fish and Wildlife and Canadian Wildlife Service 1986), Ducks Unlimited Canada (DUC), and other conservation organizations. Beginning in 1985, parcels of previously cultivated lands were seeded with perennial grasses and legumes to resemble quality waterfowl nesting habitat. Approximately 55,800 ha are currently in permanent cover in prairie Canada (Devries 2003).

Management of these parcels is necessary to mimic natural perturbations and maintain optimal nesting habitat (Naugle et al. 2000).

Aspen parklands form the northern transition between grasslands and boreal forest and have been severely altered through agriculture such that extensive tracts of native prairie are rare and the remainder has been lost or highly fragmented (Sugden and Beyersbergen 1984). While prairie species have evolved with native predators, predator abundance and community structure have changed as a consequence of landscape alteration, persecution of coyotes (*Canis latrans*), and extermination of gray wolves (*Canis lupus*; Johnson and Sargeant 1977; Sargeant et al. 1993). Control or removal of large mammalian predators can result in meso-predator release such that smaller predatory species become more numerous (Estes 1996; Henke and Bryant 1999). Red fox (*Vulpes vulpes*) populations benefit from coyote removal – as do a variety of small and medium sized mammals – due to reduced interspecific competiton (Greenwood and Sovada 1998; Henke and Byant 1999). Raccoons (*Procyon lotor*), badgers (*Taxidea taxus*), and Franklin's ground-squirrels (*Spermophilus franklinii*) have exhibited range expansions in light of anthropogenic landscape changes and coyote removal (Greenwood and Sovada 1998).

Changes in the avian community have been linked to the installation of shelterbelts, telephone and hydro-electric poles and wires, aspen encroachment, and protection from human persecution to the effect that range expansion has been reported for American crows (*Corvus brachyrhynchos*), black-billed magpies (*Pica pica*), and red-tailed hawks (*Buteo jamaicensis*; Murphy 1993; Sargeant et al. 1993). While predator communities are temporally and spatially variable, red fox, striped skunk, American

crow, and badger have been implicated as principle nest predators within the PPR (Greenwood 1986; Johnson et al. 1989). To combat severe nest depredation, predator control or removal have been used under a variety of situations but Beauchamp et al. (1996*b*) contend nest success has decreased regardless of predator management.

Of the predators affecting duck production identified by Sargeant et al. (1993), many tend to be generalists with diets consisting of small mammals, birds, birds' eggs, insects, and amphibians. Generalist predators are capable of switching from one prey item – item A (e.g. rodents) – to another prey item – item B (e.g. game bird nests) – when abundance of A is compromised – otherwise stated, in part, as the Alternative Prey Hypothesis (APH; see Angelstam et al. 1984; Angelstam 1985). Thus high vole densities may buffer predation on waterfowl nests. Most APH research involving game birds has been carried out in arctic and sub-arctic environments where predator-prey relationships are closely linked. Predator populations are hypothesized to increase – although with a lag effect – during the peak phase of a prey cycle. It is also during this peak phase that survival of alternative prey species increases due to reduced predation pressure. Reduced fox depredation on ground nests during peak phases of rodent (e.g. lemmings and voles) cycles has been reported for grouse (Angelstam et al. 1984; Wegge and Storaas 1990), geese (Summers et al. 1998; Bêty et al. 2002), and ducks (Pehrsson 1986). Similarly, avian predator diets are known to depend on rodent abundance, with decreased predation on game birds during peak rodent availability (Korpimäki et al. 1990; Reif et al. 2001; Selas 2001). However, in prairie environments, rodent populations appear to exhibit erratic fluctuations rather than cycles (Getz et al. 2001). Nevertheless, food resource availability has been shown to influence predation rates on duck nests (Byers 1974;

Crabtree and Wolfe 1988) and there is some evidence to support the APH within temperate regions (Jedrzejewski et al. 1994; Ackerman 2002). Availability of mobile prey items often depends on an array of conditions, least of which include weather, season, habitat, and predator abundance. Perturbations to prairie environments impact prey abundance such that populations should increase when optimal habitat conditions are reached. Prey populations are often at the mercy of cover management decisions aimed at improving habitat for game species.

This study was carried out in conjunction with Institute for Wetland and Waterfowl Research/DUC's Planted Cover Management Study aimed at elucidating the impacts of management – vegetation complex coupled with cover removal by means of haying or burning – on duck production. Primarily, I examined the role of alternative prey populations as they relate to duck nesting success within the constraints of managed planted cover parcels. The study design allowed for an indirect test of the second APH prediction outlined by Angelstam et al. (1984). When vole density declines, predators will select alternative prey of about the same size and accessibility as voles. Therefore, a shift to duck nests during low vole availability should result in decreased nest success and increased nest success during peak vole availability.

Overview

Organization of this thesis follows a logical progression from prominent prey use of managed habitats to the influence these populations have on duck production. To better comprehend management effects on prey communities, an examination of habitat and prey use is required and is presented in Chapter 2. With an understanding of prey availability within the constraints of managed permanent cover, prey abundance is related

to duck nest success. This is examined in Chapter 3, followed by a general discussion of analytical findings, relative concepts, and management implications in Chapter 4.

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CHAPTER 2

Effects of Planted Cover Management on Small Mammal and Beetle Abundance and Biomass within the Aspen Parkland of Saskatchewan and Manitoba[1]

Abstract. We studied relationships between management, temporal effects, vegetation, climate, landscape attributes, and small mammal and beetle abundance and biomass in the Aspen Parkland of Saskatchewan and Manitoba. Arable fields, previously converted to either a native grass-legume or a tame grass-legume stand had been subsequently managed through either burning or haying. Forty-five fields with between 1 and 8 years post-management were sampled in 2000 and 2001, 35 of which were sampled both years. In 20,500 trap nights, 1,500 small mammals were captured and 53,565 beetles were captured in 28,698 trap nights. Models including vegetation characteristics but excluding management effects were the most parsimonious for vole, shrew, and total small mammal abundance based on Akaike's Information Criterion. Year was an important parameter for shrew and total mammal abundance but not for vole and mouse abundance although the total number of small mammal captures decreased by 60% between sampling years. The greatest abundance of deer mice occurred in the first two years post-management. Abundance increased with vegetation density for all mammal groups. Beetles responded negatively to duff depth and conserved soil moisture rather than management effects although Carabid numbers were highest in the second and third years post-management. Sample-year was an important parameter in modelling the increase in beetle abundance and Carabid biomass. Overall, vegetation characteristics and annual variation in abundance appear to be better parameters than management techniques.

[1] This chapter is formatted for submission to the Canadian Journal of Zoology.

Key Words: Akaike Information Criterion, AIC_c, planted cover, Aspen Parkland, abundance, biomass, *Microtus pennsylvannicus*, *Peromyscus maniculatus*, meadow vole, deer mouse, ground beetle, Carabidae, Saskatchewan, Manitoba

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Introduction

Much of the Canadian prairie is represented by a mosaic of agricultural crops and other uses. Most native prairie habitat has been lost and the remainder is highly fragmented (Sugden and Beyersbergen 1984). Habitat loss has been ameliorated through permanent cover programs that increase nesting cover for upland nesting waterfowl species. Such programs create suitable habitat for a plethora of small mammals, birds, amphibians, and invertebrates inhabiting the prairie landscape. Waterfowl use of cover programs is well documented but the importance of cover management techniques to other wildlife is not clear. Small mammals and insects, primarily meadow voles (*Microtus pennsylvanicus*), deer mice (*Peromyscus maniculatus*), and beetles, are of interest because of their importance within the prairie ecosystem, and also for their hypothesized role as alternative prey for upland waterfowl nest predators (Byers 1974; Ackerman 2002).

Habitat conditions are a result of climate, natural disturbance, foraging by terrestrial species, and anthropogenic manipulation, the bulk of which is agricultural alteration. Small mammal populations exhibit a dynamic response to microhabitat (vegetation density, litter, and bare ground) and macrohabitat (vegetation association) variables on both spatial and temporal scales (Morris 1984; Snyder and Best 1988; Getz et al. 2001). As a result, much focus has been placed on small mammal microhabitat

selection and response to vegetative parameters. Meadow voles and deer mice often exhibit asynchronous habitat selection. Meadow voles tend to prefer moist habitats with dense vegetation but are able to persist under a range of conditions (Reich 1981; Adler and Wilson 1988; Basquill and Bondrup-Nielsen 1999*a*; Getz and Hofmann 1999). Deer mice are habitat generalists, found in low densities in most environments, with irruptions often following severe disturbances such as fire (Tester 1965). Lysne (1991) and Skinner et al. (1995) found that deer mice constituted an overwhelming majority of captures within the first few years in areas seeded to permanent cover. Skinner et al. (1995) also determined that meadow vole abundance peaked in years 3 and 4 post-seeding. We attempted to discern whether these relationships held for established permanent cover stands managed for waterfowl production.

Terrestrial beetles are a primary component of shrew diets and also constitute part of many grassland mammalian and avian diets (Martin et al. 1961). The role of beetles for control of agricultural crop pests and use in integrated pest management has great value in agriculturally dominated areas. But habitat requirements vary by species and metapopulation survival for many is on small spatial scales (Lövei and Sunderland 1996). Therefore, response to disturbance is variable depending on species, time and type of disturbance, and surrounding landscape (Holland and Luff 2000; Reed 1997). Although insect ecology has been well studied in agroecosystems, information for native and managed prairie grasslands is limited (Reed 1997).

To address the leading question of how small mammal and insect populations respond to permanent cover management techniques over time, we examined these communities on croplands previously seeded to native or tame grass/legume mixes.

Established stands had been managed through either burning or haying 1 to 8 years prior to study commencement. In addition, we collected data on project size, vegetation structure, wetland abundance, surrounding land-use, and calculated conserved soil moisture values to address environmental and landscape effects. Specifically, we sought to determine:

(1) how small mammal abundance and biomass vary among treatments and age post-management, and

(2) how beetle, particularly ground beetle (Carabidae), abundance and biomass vary among treatments and age post-management.

Materials and methods

Study Sites

Sites were primarily located within the Aspen Parkland Ecoregion of Saskatchewan and Manitoba although seven sites were located within the Moist Mixed Grassland Ecoregion of Saskatchewan (Figure 2.1) as classified by the Canadian Land Classification System (Ecological Stratification Working Group 1995). This prairie region is comprised of nearly level glacial till, glacial lake, or glacial river plains with numerous hummocky and ridged moraines throughout. Elevation ranges between 500 and 600 m with the exception of hummocky morainal deposits such as the Touchwood Hills, Saskatchewan that exceed 730 m. Humid continental climates prevail with mean July temperatures of 18.0 °C and mean January temperatures of -18.9 °C; mean annual precipitation in Saskatchewan is 420 mm (Acton et al. 1998). Dark brown and black chernozemic soils dominate the landscape with dark gray chernozems occurring in boreal transition areas and solonetzic soils occurring in association with inland drainage basins

(Acton et al. 1998). Although several major buried aquifers are present within the region, blanket aquifers are important sources of groundwater. Numerous isolated wetlands dot the landscape, ranging from ephemeral ponds, remaining wet only a few weeks in spring, to semi-permanent deep open-water ponds. Many permanent ponds and lakes are present in the region with saline lakes and ponds.

Through Ducks Unlimited Canada (DUC) and the North American Waterfowl Management Plan (NAWMP; U.S. Fish and Wildlife Service and Canadian Wildlife Service 1986), parcels of previously cultivated lands were converted to permanent waterfowl nesting cover between 1985 and 1995. Conversion consisted of seeding with either a native grass/legume mixture or an introduced (tame) grass/legume mixture. Seed mixes varied by region to accommodate soil types and moisture regimes with the more common native species being: northern, western, or slender wheatgrass (*Agropyron spp.*), green needle-grass (*Stipa viridula*), side-oats grama grass (*Bouteloua curtipendula*), sand dropseed (*Sporobolus cryptandrus*), prairie clover (*Dalea spp.*), black-eyed susan (*Rudbeckia hirta*), meadow blazingstar (*Liatrus ligulstylis*), bergamot (*Monarda fistulosa*), cut-leafed anemone (*Anemone multifida*), prairie coneflower (*Ratibida columnifera*), hedysarum (*Hedysarum spp.*) and western snowberry (*Symphoricarpos occidentalis*). Introduced mixes may include tall, intermediate, or slender wheatgrass (*Agropyron spp.*), wild rye grass (*Elymus spp.*), brome grass (*Bromus spp.*), creeping red fescue (*Festuca rubra*), and alfalfa (*Medicago spp.*). Stands tended to exhibit a high degree of structural diversity and heterogeneous density within and between fields.

Projects were managed post-seeding to control weed and shrub invasions and to maintain/improve stand health and vigour. For the purposes of the study, projects

managed 1 to 8 years prior to this study through either burning or haying were included. No management occurred subsequent to these burn/hay events. Projects were categorized by treatment (TRT) as native-burn, native-hay, or tame-hay with an age (i.e. growing seasons) since management identifier (year post-management [YPM]). For the most part, DUC does not manage tame stands through burning, therefore, there was no tame-burn treatment. YPM categories were not constant within fields among years since stand age increased by one year between sampling years (Figure 2.2). YPM 6 to 8 were pooled into one 6+ YPM category because there were few or no replicates within treatments. Forty-one projects were sampled in 2000; 42 were sampled in 2001. Project sizes (area of planted cover) ranged from 6 to 62 ha (refer to Appendix A for project characteristics).

Geographic and Climatic Characteristics

Geographical location (latitude and longitude) and conserved soil moisture (CSM) were included as environmental variables. Project latitude and longitude, in the form of decimal degrees were taken from the DUC Project Database. CSM was used as an index of wetness regimes and wetland conditions (Boyd 1981; Bethke and Nudds 1995) where precipitation records for the 21 months preceding May 1, 2000 and 2001 were used to calculate soil moisture according to the Williams and Robertson (1965) formula:

$$CSM = 0.36A+[0.37B–0.2(0.36A)]+0.13C+\{0.30D–0.2[0.36A+(0.37B–0.2(0.36A))+0.13C]\}$$

where A = total precipitation during August through October in year $t - 2$;

B = total precipitation during November in year $t - 2$ through April in year $t - 1$;

C = total precipitation during May through October in year $t - 1$;

D = total precipitation during November in year $t - 1$ through April in year t.

Prior to calculating CSM, a proximity analysis using ArcGIS (Environmental Systems Research Institute, Redlands, California) software was conducted to identify weather stations within 30 km of each project. We used monthly precipitation records from 50 stations and calculated CSM per weather station for 2000 and 2001 (Monthly Precipitation Records, Atmospheric Environment Service, Downsview, Ontario). Individual project CSM was generated as an average of the CSM for the three or four nearest weather stations, taking into consideration the weighted inverse distance from the project to each station.

A natural precipitation gradient occurred across the region such that moisture, and subsequently CSM, increased in a south-eastwardly direction. We tested for correlation between CSM and project location (latitude and longitude) with a Pearson Correlation Matrix. A marked difference in precipitation was noticed between 2000 and 2001 therefore paired t-tests were used to test for between-year difference in CSM.

Adjacent Land Use Characteristics

Land use was characterized for the 4 quarter sections adjacent to each project. Areas were visited in July, categorized to 1 of 13 land uses (see Appendix B for land use categories), and delineated on aerial photographs. Grain crops were identified as either fall- or spring-seeded whereas other cereal or oil crops were identified to crop type (i.e. flax, corn, pea, or canola). Non-cropped areas were categorized as native areas left idle, native areas with grazing, planted cover, or hay fields. Fallow areas (bare soil) and areas with buildings or recreational infrastructure (e.g. farms, churches, graveyards, or golf courses) were categorized separately from surrounding land uses. Air photos were

digitized by using AutoCad (AutoDesk, Inc., San Rafael, California) software to determine aerial extent of each use category.

Land use characteristics were consolidated into two functional land use groups: IDLE (non-cropped areas) and CROP (cropped or fallow areas). Percent area per group was calculated and further arcsine transformed (Table B.24, Zar 1999) to accommodate underlying normality assumptions (Zar 1999). CROP was chosen to include in modelling because landscape-level crop information is easier to quantify and readily available through Agriculture Statistics Division of Statistics Canada.

Planted Nesting Cover Area

The amount of planted nesting cover (also referred to as dense nesting cover [DNC]) was variable between projects. To determine the area of coverage per project, the extent of nesting cover was delineated on air photos and area calculated through digitizing using AutoCad.

Wetland Area

Wetlands within each project and a circumscribed 400 m strip were classified according to Stewart and Kantrud (1971) in July of each year. Wetlands were delineated on aerial photographs during classification based on extent of wetland vegetation. There was no perceived change in the extent of wetland vegetation between sampling years, therefore, wetland area was determined through digitizing the 2000 data. Only wetlands with permanency class III, IV, and V (seasonal, semi-permanent, and permanent) were included in the analysis.

Wetland size, density, and distribution are variable across the prairie region. To provide an index of wetland area (WA) per project, total wetland area was divided by the

area of DNC within the project. WA was natural log transformed (ln[x+1]) to more closely represent a normal distribution. Correlation between WA and CSM was tested with a Pearson Correlation Matrix.

Vegetation Characteristics

Vegetation at all projects was sampled three times (May 22 – May 28; June 12 – June 18; July 3 – July 9) per year at permanently established stations. The number of sampling stations depended on project size: 9 stations for DNC less than 32 ha, and 16 stations for DNC exceeding 32 ha. Locations of sampling stations (Universal Transverse Mercator [UTM] projection) were randomly generated using SPANS GIS (PCI Geomatics, Richmond Hill, Ontario). A grid of systematic random points was established with a Global Positioning System (GPS) throughout each project. To reduce station-placement bias, once at the approximate point, a random number of paces between 1 and 20 and direction (0 - 360°) were generated on a standard mathematical calculator. A spike was driven into the soil at the sampling point and a willow stake set 4 m north to aide in station relocation. Both spike and willow stake were flagged with orange surveyors tape.

Sampling sessions consisted of four sets of measurements at each station: visual obstruction readings (VOR; Robel et al. 1970), duff depth (DUF), maximum height (MXHT), and vegetation composition. VOR was estimated by placing a 14 dm pole demarcated at 0.5 dm intervals at the sampling point and recording the last mark not visible through the vegetation from 1 m above ground level at a distance of 4 m from the pole. Four VOR were recorded, one in each cardinal direction. Duff depth, described as the dead and/or decaying horizontal vegetative matter on the soil surface, was measured

30 cm from the sampling point in each cardinal direction. A plastic ruler was inserted vertically through the duff layer until the ruler contacted the soil layer; duff depth was recorded to the nearest 0.5 cm. Maximum vegetation height was determined as the tallest piece of new or residual vegetation within a 15 cm radius from the Robel pole. A slotted clear plastic disc (30 cm diameter) was slid down the pole until the first piece of vegetation was encountered; height was measured to the nearest 0.5 dm. Vegetation composition was measured as comparative percent grass, forb, and shrub aerial cover within one square metre.

Recently managed projects initially had substantially less cover than projects idled for several years. Seasonal growth rates were not similar and, as such, did not show parallel density trajectories throughout the summer sampling periods. We opted to use an overall summary of conditions by averaging the three sampling period means to generate an index of field-level conditions. Because percent grass and percent forb composition were ipsitive, only forb cover was included in the analysis. Percent forb composition (FORB) was arcsine transformed to accommodate underlying normality assumptions (Table B.24, Zar 1999). Percent shrub was negligible and was not included in the analysis.

Beetle Abundance

Invertebrates were sampled from May 3 – July 14, 2000 and May 7 – July 12, 2001 with pitfall traps (modified design based on Morrill 1975 and Schmidt and Lockwood 1992). Traps were located 5 m north of the odd-numbered permanent vegetation sampling stations in each project and serviced every 7 to 10 days. Traps consisted of an outer plastic beverage cup (473 ml; perimeter 27.5 cm) with a removable

inner cup trimmed down to a height of 4.5 cm. They were placed with the lip flush with ground level and the inner liner ¾ filled with a saturated salt solution (Figure 2.3). During servicing, traps were emptied, rinsed with water and re-filled with solution, or replaced if damaged. Samples within each field were combined at the time of collection and identified to Order. Specimens were stored in 70% isopropyl solution.

Beetle specimens were sorted, identified to Family, counted, and air dried. Weights were determined per Family. Voucher specimens identified by Dr. George E. Ball, Entomologist, are housed at the Strickland Museum of Entomology, University of Alberta.

Although 11 arthropod Orders were represented, only beetles were included in the analysis as they constituted the single largest percentage of captures (over 40%). Pitfall traps reflect activity and density of surface-dwelling insects, but captures are also a function of microclimate, trap material, preservative used, and beetle assemblage (Southwood 1978; Spence and Niemelä 1994; van den Berghe 1992; Waage 1985). Beetle catch per unit effort (CPUE) was calculated as total captures per 100 trap nights (TN) to determine an index of relative abundance.

$$CPUE = (A/TN) \cdot 100$$

where A = number of individuals.

Silphids (burying and carrion beetles) and Carabids (ground beetles) comprised the bulk of captures. However, Silphids are attracted to odours emitted from carrion, therefore, traps already containing captured individuals may have acted as a lure. As such, Silphid counts and weights were excluded from abundance and biomass calculations. Relative abundance was generated separately for Carabids.

Beetle Biomass

Numbers of individuals reflect density but do not provide qualitative accounts of production capacity since individuals and Families vary greatly in size. Indices of beetle biomass (excluding Silphidae) and Carabidae biomass were calculated as total dry weight of individuals per square meter (Brower et al. 1998).

$B = \sum W/A$

where $\sum W$ = sum weight of individuals (grams);

A = total trap area sampled (m^2; total number of TN•area of each trap [56.745 cm^2] /10,000). Carabidae and total beetle abundance and biomass estimates were natural log transformed (ln[x+1]) to accommodate normality assumptions. A Pearson Correlation Matrix was used to determine the degree of correlation between abundance and biomass estimates.

Small Mammal Abundance

A single trapping session for small mammals was conducted between July 16 and August 12, 2000 and July 8 and August 4, 2001. Small mammals were sampled at 25 sampling stations in a 5 X 5 grid layout (Figure 2.4) (Innes and Bendell 1988; Jones et al. 1996). Sampling stations, consisting of two Museum Special snap-traps (Woodstream Corp., Lititz, Pennsylvania), were spaced 10 m apart. Grid placement was randomly generated for each field with the selection constraint that adjacent habitats (i.e. wetlands, tree stands, crops, and roads) were more than 50 m from any side of the grid. The northwest corner of each grid was marked via GPS units to facilitate grid re-establishment in 2001. Traps were baited with a peanut butter/rolled oat mixture and serviced for five consecutive mornings (between 0600 and 1200 hrs). Due to the distance

between projects, between 9 and 12 projects were sampled during a trapping period with these traps relocated to new sites at the close of the five night trapping period.

Specimens were stored in 120 ml Whirl-Pak® bags; large individuals (e.g. *Spermophilus spp.* and *Thomomys spp.*) were stored in Zip-Lock® bags. Date, location, species, and sex (if determinable) were recorded at the time of collection. Confirmation of species identification, sex, breeding condition, weight, and standard body measurements were determined at the Museum of Zoology, University of Alberta where the specimens are stored. Sampling procedures were approved by the Manitoba Wildlife Animal Care Committee (MWACC #2000-01) and the Faculty of Agriculture, Forestry and Home Economics Animal Policy and Welfare Committee (FAPWC #2001-10D).

A measure of relative abundance (number of small mammals caught relative to the trapping effort) was generated by calculating corrected catch effort (CE) as a measure of trap success per 100 TN (Nelson and Clark 1973; Holroyd and Van Tighem 1983; Beauvais and Buskirk 1999). Relative abundance estimates assume capture numbers represent a relatively constant but unknown relationship to the total population size although the number of captures further depends on animal activity, movement, and trapper skill (Krebs 1994). If these conditions are reflected in trapping success, catch effort becomes an index of abundance measuring relative density of animals. CE was calculated following the Nelson and Clark (1973) formula that accounts for decreases in trapping efficiency due to traps being sprung by animals other than the species of interest, weather, vegetation, or any other cause.

$$CE = (100 \cdot A)/(TU-S/2)$$

where A = number of individuals of the desired species caught;

TU = P•I•N (number of trapping units);

P = number of trapping intervals;

I = length of trapping interval;

N = number of traps;

S = number of traps sprung by all causes.

Non-mammalian specimens were included in the analysis only as sprung traps to determine corrected CE. Due to low captures of some species, individuals were grouped together as functional units such that vole species (*Microtus pennslyvanicus* and *Clethrionomys gapperi*) were grouped and shrew species (*Sorex haydeni*, *Sorex cinereus*, *Sorex hoyi*, *Sorex arcticus*, and *Blarina brevicauda*) were grouped. Pocket gophers and thirteen-lined ground squirrels were considered incidental captures as they were non-target species. Jumping mice occurred in too few numbers and were not included in the analysis. Total small mammal CE was generated by pooling vole, deer mice, and shrew captures. Deer mice abundance data were reciprocal (1/[x + 1]) transformed while vole, shrew, and total abundance data were natural log transformed (ln[x + 1]) to accommodate normality assumptions.

Small Mammal Biomass

Biomass, as a quantitative measure of community-level production, was calculated per species group (deer mice, voles, and shrews) and total small mammal biomass as mean wet weight (kg) of individuals per hectare (Brower et al. 1998).

$B = \sum W/A$

where $\sum W$ = sum weight of individuals (grams)/1000;

A = total area sampled (area of trapping grid = 1600 m^2 or 0.16 ha).

Biomass calculations were transformed with a reciprocal (1/[x+1]) transformation for deer mice and natural log (ln[x+1]) transformation for voles, shrews, and total small mammals. A Pearson Correlation Matrix was used to determine the degree of correlation between abundance and biomass estimates.

Analysis

Data were tested for normality prior to analysis, transformed where appropriate, and tested for correlation. The alpha level was set at 0.05 for all tests. Pairwise comparisons of the least square means were conducted with the overall error rate controlled by Tukey's multiple comparisons adjustment with alpha set at 0.05. A split-plot type analysis within SAS' PROC MIXED was used because YPM categories increased by 1 YPM between sampling years while treatment remained the same (Littell et al. 1996; SAS Institute 1999). In addition, there was uneven replication or missing data within cells. Given the variability in seed mixes and site location, it is possible that projects within one management regime exhibited more variability than another regime. To accommodate for heterogeneity in error variance, differences between plots within a treatment regime (treatment factor) were captured by between-plot variances and changes from 2000 to 2001 (temporal factor) within a project were captured by within-plot variances. Data were initially fit with four alternative variance structures:

(1) homogenous variances within treatment regimes and between sampling years,

(2) variances within treatment regimes varying by treatment regime; homogenous variances between sampling years,

(3) homogenous variances within treatment regimes; between sampling years variance varying by treatment regime, and

(4) both within treatment regime and between sample-year variance varying by treatment regime.

Restricted/residual maximum likelihood estimation method was employed in this process because of the need to estimate the variance components (Verbeke and Molenberghs 2000). The most parsimonious model structure was chosen based on Akaike's Information Criterion (AIC_c), adjusted for small sample size because the ratio of sample size to parameters was less than 40 (Akaike 1973; Burnham and Anderson 1998):

$$AIC_c = (-2 \log likelihood) + 2K + (2K[K+1] / [n-K-1])$$

where log likelihood is the maximized log-likelihood value, K is the total number of estimated parameters (including all independent variables, dependent variables, intercept, random factor, and residual variance), and n is sample size. The model with the lowest AIC_c is considered the most parsimonious or "best" model providing a quantitative assessment of the "strength of evidence" given the data (Burnham and Anderson 1998). Inferences were made on candidate models in which the ΔAIC_c (AIC_c - minimum AIC_c) values were ≤ 4 (Burnham and Anderson 1998).

Using the chosen variance model structure, models were refit with the maximum likelihood estimation method which provides estimators of the fixed effects (Verbeke and Molenberghs 2000). A suite of 10 *a priori* hypothesized models were examined for their ability to predict VOR, DUF, MXHT, and FORB response to management parameters. YR, TRT, and YPM were included because vegetation characteristics are a function of seed mix, treatment, and stand age. A null model, which used only the intercept, random, and residual parameters to model the response variable, was included in each model suite as a measure of model fitness. Inferences were based on models with ΔAIC_c values ≤ 2

(Burnham and Anderson 1998). Akaike model weights were used to determine the weight of evidence for each model (Burnham and Anderson 1998):

$$w_i = \exp(-0.5 * \Delta_i) / \sum^{R}_{r=1} \exp(-0.5 * \Delta_i)$$

where $\exp(-0.5 * \Delta_i)$ is the relative likelihood of a model and $\sum^{R}_{r=1} \exp(-0.5 * \Delta_i)$ is the sum of all relative likelihoods within the candidate set of models. An assessment of model strength relative to other models was achieved using model weight ratios (w/w_i) where the best model has a weight ratio of 1. To quantify the importance of each variable over the set of competing models, average variable weights (w_v) for each variable were calculated by summing the model weights in which a specific variable occurs and dividing it by the number of models in which it occurs (Burnham and Anderson 1998).

Non-correlated total beetle and Carabid abundance and biomass estimates were first fit to the management suite of 10 models as above (refer to Appendix C for model construction). Data were then fit to 15 *a priori* vegetation models to assess the response to VOR, DUF, MXHT, and FORB. Because relationships may not have been linear, quadratic vegetation covariates were included in the *a priori* vegetation suite. As project features may also influence local populations, linear and quadratic landscape covariates (CSM, CROP, WA, and DNC) were modelled to determine individual importance. The best models from each suite (ΔAIC_c values ≤ 4) were pooled together and exploratory models developed from covariate combinations. Inferences were based on models from the exploratory suite with ΔAIC_c values ≤ 2. As outlined above, Akaike model weights and average variable weights were also considered.

Non correlated small mammal abundance and biomass estimates were fit to *a priori* management and vegetation models, landscape covariates, and exploratory models

as outlined for beetle data. Due to the possibility that competition between voles and mice, or predation on vole nests by shrews, may exist, vole, mice, and shrew abundance estimates were also included as landscape covariates. Beetle and Carabid abundance were included as landscape covariates because of their importance in shrew diets. Statistical analysis was performed with SAS, version 8.2.

Results

Unless otherwise indicated, the homogenous variance structure (Structure 1) was chosen as the best structure and employed for final analysis.

Geographic and Climatic Characteristics

CSM exhibited strong negative correlations with latitude ($r = -0.848$) and longitude ($r = -0.872$) of study site locations. An increase in CSM occurred with decreased latitude and increased longitude. In other words, locations within the south-eastern portion of the study site (e.g., the Minnedosa region of Manitoba) had higher moisture levels than the north-western portion (e.g., the Saskatoon area). There was a marked decrease in CSM such that emerging drought conditions in Saskatchewan were apparent during the 2001 summer field season ($t_{0.05(2),35} = 4.93$, $P < 0.0001$). Although CSM in Manitoba was lower in 2001, Manitoba received abundant precipitation during the summer period (May – August) which is not reflected in the CSM calculation.

Wetland Area

Wetland area was not correlated to CSM ($r = 0.023$), therefore, both variables were included in the analyses.

Vegetation Characteristics

Although three models were within 0.5 ΔAIC_c units, the most parsimonious model, YR+TRT+YPM, indicated VOR differed by year, treatment type, and YPM (Table 2.1). Density decreased by 20% between sample-years but was lowest within Native-burn at 1.5 dm (SE = 0.1034) and greatest within Tame-hay at 2.0 dm (SE = 0.1045; Figure 2.5). Readings were lowest in the first YPM, highest in YPM 2 and 3, and gradually decreased in subsequent years (Figure 2.6). Average variable weights revealed that year (w_v = 0.1000) was almost twice as important as the next best variable, YPM (w_v = 0.0573). Given that both CSM and VOR measures decreased between sampling years, vegetation density was re-examined with management and CSM parameters to determine the effect of CSM on density. TRT+CSM and YPM+CSM were competing models, indicating CSM affected density measures. The TRT+CSM model weight (w/w_i = 0.4686) was twice that of YPM+CSM (w/w_i = 0.2407), suggesting CSM impacts within treatments were more pronounced. The effect of CSM on VOR within treatments is shown in Figure 2.7 where the magnitude of change varied among treatment types. VOR did not increase with CSM in Native-burn projects as it did in Native-hay and Tame-hay projects.

Variance Structure 3 was selected for duff depth. Duff depth varied by treatment type with Native-burn projects exhibiting a shallower duff layer (0.9 cm, SE = 0.1) than Native-hay and Tame-hay (1.2 cm [SE = 0.08] and 1.6 cm [SE = 0.17] respectively; Figure 2.8). CSM in association with TRT provided a better fit than TRT alone (Table 2.1). This suggests CSM was explaining some of the variation among treatments such that duff was thicker in Tame-hay projects and increased moreso in relation to CSM than other treatments (Figure 2.9).

Maximum height was best modelled with YR+YPM (Table 2.1). There was a decrease in mean MXHT between sample-years with values increasing after the first YPM then stabilizing in subsequent years (Figure 2.10).

FORB was lowest within Native-Burn projects with 13% compared to 51% and 33% for Native-Hay and Tame-Hay projects respectively (Figure 2.11). TRT and YR+TRT models best fit the data and exhibited nearly equal model weight ratios (w/w_i = 1 and w/w_i = 0.9620 respectively; Table 2.1). While there were pronounced differences in forb cover among treatments, a slight increase in forb composition (5%) between sample-years further explained the variability.

Beetle Abundance

Of the 13 Coleopteran Families represented by captures, Silphidae and Carabidae constituted the majority of individuals with 52% and 37% respectively (Table 2.2). When Silphidae were excluded from measures, Carabidae constituted 78% of captures and 85% of biomass (range 40 – 100%). Twenty-eight Carabid species are represented among the 56 species identified (refer to Appendix D for the list of species). Total beetle abundance and biomass were highly correlated (r = 0.82), therefore, only total abundance models are presented. Carabid abundance and biomass models are presented because the estimates were not as highly correlated (r = 0.61).

Total beetle relative abundance ranged from 89 (95% CI = 132, 60) to 121 (95% CI = 222, 65) captures/100 TN in the first 3 YPM, decreasing to 47 (95% CI = 69, 32) to 62 (95% CI = 84, 46) captures/100 TN in subsequent YPM. There was little difference in mean abundance among treatments (range 66 [95% CI = 88, 49] to 75 [95% CI = 103, 53] captures/100 TN). However, management parameters cannot be considered to

adequately model abundance because only the YR model resulted in a better fit than the null model (Table 2.3). More beetles were captured in 2001 (85 beetles/100 TN, 95% CI = 106, 68) than 2000 (58 beetles/100 TN, 95% CI = 72, 46). Examination of vegetation models revealed the DUF+FORB model to be the most parsimonious. A negative relationship existed between abundance and DUF ($r^2 = 0.18$; $\beta_{DUF} = -0.485$, SE = 0.11; Figure 2.12) while abundance appeared to increase with FORB ($r^2 = 0.1156$; $\beta_{FORB} = -0.017$, SE = 0.005; Figure 2.13). Of the landscape covariates, total abundance fit a quadratic CSM model better where captures were greatest in the intermediate CSM range ($r^2 = 0.1129$; $\beta_{CSM} = 0.054$, SE = 0.033; $\beta_{CSM}^2 = -0.00023$, SE = 0.00012; Figure 2.14). Twenty-four models were developed from combining the better (ΔAIC_c values ≤ 4) management, vegetation, and landscape covariates. The most parsimonious model included YR, DUF, FORB, and quadratic CSM terms (Table 2.3). Average variable weights indicated that the DUF+FORB pair was the most important variable ($w_v = 0.0833$) with quadratic CSM ($w_v = 0.0570$) and YR ($w_v = 0.0413$) relatively less important. Total beetle abundance appears to be best explained through duff depth and forb cover although CSM and between-year difference further explained some of the variability.

Although Carabids were most abundant in YPM 2 and 3 with 78 captures/100 TN (95% CI = 112, 54) and 83 captures/100 TN (95% CI = 121, 58) respectively, decreasing to 41 captures/100 TN (95% CI = 54, 31) in years 6+, management effects did not adequately explain Carabid abundance since the null model was the most parsimonious of the management models (Table 2.4). As with beetle abundance, there was very little difference among treatments. Of the vegetation models, linear and quadratic DUF

models were the better models with model weight ratios of $w/w_t = 0.9406$ and $w/w_t = 1$ respectively, almost twice the model weight of the nearest competitor. Abundance appeared to decrease with greater duff depth, stabilizing after approximately 1.5 cm ($r^2 = 0.1164$; $\beta_{DUF} = -0.946$, SE = 0.408; $\beta_{DUF}^2 = 0.199$, SE = 0.125; Figure 2.15). Six landscape models were within 2.0 ΔAIC_c units, one of which was the null model, indicating that these models could not predict Carabid abundance well. A weak relationship existed between CSM and Carabid abundance where numbers were highest at intermediate levels of CSM ($r^2 = 0.0518$; $\beta_{CSM} = 0.0768$, SE = 0.0322; $\beta_{CSM}^2 = -0.00028$, SE = 0.00012; Figure 2.16). Exploration with YR and YPM, plus vegetation and landscape models (147 candidate models in the suite) revealed one superior model which included YPM with quadratic VOR and CSM terms. Carabid abundance appeared to be lowest at intermediate levels of VOR ($r^2 = 0.0886$; $\beta_{VOR} = -0.2.227$, SE = 0.656; $\beta_{VOR}^2 = 0.498$, SE = 0.175; Figure 2.17). With CSM and VOR explaining most of the variation (average variable weight ratios of $w_v = 1$ and $w_v = 0.7964$ respectively), differences among YPM were more pronounced. Abundance (antilog[x]-1of the least square means) was considerably less in the first YPM at 58 captures/100 TN (95% CI = 99,34), highest in the second (97 captures/100 TN, 95% CI = 135, 69) and third YPM (90 captures/100 TN, 95% CI = 125, 65), then decreased in subsequent YPM to a low of 37 captures/100 TN (95% CI = 47, 29) in YPM 6+ (Figure 2.18).

Carabid Biomass

Similar to Carabid abundance, biomass was slightly greater in the first 3 YPM but YR was the most parsimonious management model (Table 2.5). Mean biomass doubled from 1.6 g/m^2 (95% CI = 2.1, 1.3) to 3.0 g/m^2 (95% CI = 3.7, 2.4) between 2000 and

2001. There was a difference of 0.1 g/m^2 among treatments. Three *a priori* vegetation models were within 2.0 ΔAIC_c units with FORB and DUF having nearly equal average variable weights of $w_v = 0.1262$ and $w_v = 0.1187$ respectively. There was a positive relationship between Carabid biomass and forb composition ($r^2 = 0.0824$; $\beta_{FORB} = 0.011$, SE = 0.00447; Figure 2.19); a negative relationship existed between duff depth and Carabid biomass ($r^2 = 0.1628$; $\beta_{DUF} = -0.346$, SE = 0.0894; Figure 2.20). The quadratic CSM model was the best landscape covariate model such that Carabid biomass appeared to decrease when CSM exceeded 130 mm ($r^2 = 0.1224$; $\beta_{CSM} = 0.0433$, SE = 0.0254; $\beta_{CSM}{}^2 = -0.00018$, SE = 0.000093; Figure 2.21). The combination of YR+DUF+CSM+CSM^2 provided the most parsimonious model with twice the model weight ($w/w_i = 0.4616$) than the nearest competitor ($w/w_i = 0.2243$). Elevated biomass estimates were apparent in 2001 but biomass decreased overall with greater duff depth and conserved soil moisture.

Small Mammal Abundance

A total of 10,000 trap nights were generated in 2000 with 1,038 small mammals captured; 10,250 trap nights were generated in 2001 with 462 captures. The most commonly captured species were meadow voles (*Microtus pennslyvanicus*; 61%), deer mice (*Peromyscus maniculatus*; 14%), and masked shrews (*Sorex cinereus*; 13%) (Table 2.6). Incidental non-mammalian captures included 17 sparrows (Bairds' [*Ammodramus bairdii*]; savannah [*Passerculus sandwichensis*]; clay-colored, [*Spizella pallida*]), 3 tiger salamanders (*Ambystoma tigrinum*), 2 garter snakes (*Thamnophis spp.*), and 1 wood frog (*Rana sylvatica*). Abundance and biomass estimates were highly correlated (r = 0.84 to r = 0.98), therefore, only abundance models are presented. Because beetle abundance and

biomass estimates were highly correlated, total beetle abundance was included as an index of food availability within the landscape features model suite.

Vole abundance was lowest in Native-Burn projects at 2.1 captures/100 TN (95% CI = 3.2, 1.3) and highest in Native-Hay and Tame-Hay projects at 4.3 captures/100 TN (95% CI = 6.6, 2.8) with more captures in YPM 2 through 4 (over 4.1 captures/100 TN; 95% CI = 7.7, 2.1) and lowest in the first YPM (1.2 captures/100 TN; 95% CI = 3.7, 0.3). However, the YR model carried two times the model weight ($w/w_i = 0.6620$) of the nearest competitor, YR+TRT ($w/w_i = 0.3100$; Table 2.7). This suggests that while treatment type may have some explanatory power, yearly variation, in this case a 55% decrease in capture numbers, was the better explanatory variable among management effects. Five vegetation models were within 4 ΔAIC_c units with the VOR+DUF model selected as the most parsimonious. VOR and vole abundance ($r^2 = 0.1676$; $\beta_{VOR} = 0.593$, SE = 0.169) showed a positive relationship (Figure 2.22) but because the VOR+DUF pair is a better model than VOR alone, duff depth was imparting additional information. A weaker positive relationship existed between duff depth and vole abundance ($r^2 = 0.0947$; $\beta_{DUF} = 0.294$, SE = 0.148; Figure 2.23).

Fitting landscape covariates resulted in two shrew abundance models being considered with the linear model having three times the model weight. Vole abundance increased in unison with shrew abundance ($r^2 = 0.1834$; $\beta_{SHRW} = 0.631$, SE = 0.148; Figure 2.24). Covariates and covariate pairs within 4 ΔAIC_c from each model suite (management, vegetation, and landscape) were combined for further exploration. Of the 54 exploratory models, four had $\Delta AIC_c < 2$ and are presented in Table 2.7. The two best models, VOR+DUF+SHRW and VOR+MXHT+SHRW held equal model weights and

considerably more strength ($w/w_i = 1$) relative to the next competitor, VOR+SHRW ($w/w_i = 0.65$). Average variable weights indicated there was more evidence to suggest the vegetation pairs ($w_v = 0.0326$ and $w_v = 0.0321$ for VOR+DUF and VOR+MXHT respectively) were better predictors relative to SHRW ($w_v = 0.0255$). Voles appeared to increase simultaneously with shrews but were most likely responding to vegetation density in association with duff depth or maximum height parameters.

The YPM model was the best *a priori* management model to fit deer mice abundance (Table 2.8). Deer mice were most numerous in the first 2 YPM (1.2 individuals/100 TN [95% CI = 3.3, 0.5] and 1.4 individuals/100 TN [95% CI = 2.6, 0.7] for YPM 1 and 2 respectively) with abundance falling to 0.3 individuals/100 TN (95% CI = 0.7, 0.2) in YPM 3 and remaining low in subsequent years (Figure 2.25). Eight vegetation models were within 4.0 ΔAIC_c units but VOR had twice the model weight as the nearest competitor. VOR was present in all top models although there was only a difference of 4.78 ΔAIC_c units between the null and best models. VOR had a weak negative relationship with the reciprocal of deer mouse abundance indicating an increase in abundance with increased vegetation density ($r^2 = 0.0632$; $\beta_{VOR} = -0.164$, SE = 0.0601; Figure 2.26). Although DNC was the better landscape model, it was only slightly better than the null model, indicating landscape covariates by themselves were poor predictors of mouse abundance. Combinations of YPM and YR+YPM with vegetation and landscape covariates from models within 4.0 ΔAIC_c units resulted in 458 exploratory models. Three models, YPM+VOR+VOLE, YPM+VOR+CARB, and YPM+CARB, of nine competing models were within 0.41 ΔAIC_c units with considerably larger model weight ratios relative to the nearest competitor (Table 2.8). Examination of the average

variable weights revealed vole abundance as the single most important variable, followed in order of weight by Carabid abundance, YPM, and VOR. However, only weak negative relationships existed when mouse abundance was regressed separately with vole abundance ($r^2 = 0.0286$; $\beta_{VOLE} = 0.0228$, SE = 0.0353; Figure 2.27) and Carabid abundance ($r^2 = 0.0057$; $\beta_{CARB} = 0.041$, SE = 0.0461). When the variables were combined with YPM and each other, the models provided a better fit. A negative relationship between vole abundance and the reciprocal of mouse abundance implies that as vole numbers increased, deer mice decreased. The negative effects of increased vole abundance and/or Carabid abundance, in conjunction with vegetation density and stand age, appeared to help explain mouse abundance being greatest in the first two YPM.

Similar to voles, sample-year was the best explanatory variable of shrew abundance as there was an 83% decrease in shrew captures between years (Table 2.9). The number of captures did not differ among treatments or YPM. One vegetation model, VOR+FORB, was superior to the rest with shrew abundance increasing slightly with vegetation density ($r^2 = 0.0939$; $\beta_{VOR} = 0.479$, SE = 0.115; Figure 2.28) but decreasing slightly with forb composition ($r^2 = 0.0557$; $\beta_{FORB} = -0.0151$, SE = 0.00458; Figure 2.29). Linear and quadratic vole abundance models were selected as competing landscape models where the linear model had a stronger weight. The positive relationship between vole abundance and shrew abundance was previously shown during discussion of vole abundance. Linear and quadratic beetle covariates, although they were not deemed good predictors by themselves, were included in the exploratory suite because beetles make up a large proportion of shrew diets. Examination of 36 exploratory models revealed 14 competing models with the YR+VOR+FORB+VOLE model being the most

parsimonious. Due to the large number of competing models, average variable weights were assessed. YR held a greater average variable weight (w_v = 0.0555) relative to the remaining covariates which held average weights between w_v = 0.02478 and w_v = 0.0292, indicating shrew abundance was highly variable between years but that vole abundance, beetle abundance, vegetation density, and forb cover were also imparting additional information. A weak negative relationship existed between beetle abundance and shrew abundance (r^2 = 0.088; β_{BEET} = -0.244, SE = 0.0870).

Voles constituted the bulk of captures, therefore total small mammal abundance patterns resembled vole abundance patterns. Relative abundance per mammal group compared to total small mammal abundance among YPM is shown in Figure 2.30. The number of animals captured was greatest in YPM 2 through 4 with the YR+TRT model being selected as the most parsimonious management model (Table 2.10). Total abundance was lowest in Native-Burn (3.9 captures/100 TN, 95% CI = 5.2, 2.9) and highest in Tame-Hay projects (7.6 captures/100 TN, 95% CI = 10.0, 5.6; Figure 2.31). There was a 67% decrease in captures between sample-years. Five vegetation models were within 2.2 ΔAIC_c units but VOR was the best model and held a larger average variable weight as it was present in all top models. Figure 2.32 depicts the positive relationship between vegetation density and small mammal abundance (r^2 = 0.292; β_{VOR} = 0.824, SE = 0.143). Of the landscape covariate models, the null model was within 1.40 ΔAIC_c units, suggesting that while the quadratic Carabid abundance, quadratic CSM, and total beetle abundance covariates were the better landscape predictors, they could not adequately explain small mammal response. Exploratory models were developed from the better management and vegetation models presented in Table 2.10 plus 9 landscape

covariates for a total of 181 candidate models. Sixteen models were selected as competing models, suggesting that a variety of covariate combinations are potential predictors of total small mammal abundance. However, through examining average variable weights, the VOR+FORB pair had nearly twice the variable importance (w_v = 0.01872) than YR (w_v = 0.0111) or VOR (w_v = 0.01102), suggesting that while abundance can decrease drastically between years, small mammals were primarily responding to vegetation density and forb composition.

Discussion

Planted cover vegetative characteristics are a reflection of seed mixture, stand age, previous management events, and climate (Higgins and Barker 1982). There were obvious vegetative dissimilarities between year and treatments with few differences between YPM. Reduced density, maximum height, and forb composition between sampling years was partially a function of emerging drought conditions, however, YR was a better explanatory variable than CSM, indicating CSM could not fully explain the decreases. Vegetation density, duff depth, and forb composition were all lowest in Native-Burn projects. Native-Burn projects had the least amount of forb cover and in conjunction, the lowest density, while Native-Hay and Tame-Hay projects varied only slightly in density and forb composition. This was most likely due to the greater use of forbs (e.g. alfalfa) in these seed mixes. Density increased with CSM, especially when a higher percentage of forbs were present. This can be seen by the increase in VOR with CSM in hay treatments and the lack of response in Native-burn projects.

Vegetation mix, in association with CSM, affected duff thickness. A shallow duff layer in Native-Burn projects was expected since burning removes the litter layer whereas other treatments had established duff layers. Use of the Model 3 variance structure allowed for duff depth to vary between years within a project. This model structure was a better fit than other structures and suggests that perhaps decomposition rates and litter build-up are highly variable across the region. Duff thickness tended to increase with CSM but at varying degrees among treatments. Accumulation was slower in Native-Burn projects, possibly due to a less pronounced vegetative response to CSM for native mixes compared to tame. Because more vegetative matter is available in Tame mixes, greater moisture may enhance the difference between duff depth in Tame and Native mixes. Lower vegetation density and height in the first YPM is a reflection of the lack of residual vegetation. Measurements taken in May, and to some extent June, were capturing residual vegetation characteristics plus new growth.

Overall, it appears that beetles responded to environmental parameters as opposed to management techniques with a large degree of annual variation. Catch per unit effort and biomass increased between sample-years with duff depth and conserved soil moisture further explained total beetle and Carabid biomass responses. While abundance and diversity are thought to increase with litter depth through enhanced microsite conditions (Niemelä and Spence 1994), we saw a decrease in abundance and biomass with increased duff. Similarly, Guillermain et al. (1997) found negative relationships between litter depth and beetle abundance and richness, while Molnár et al. (2001) found diversity decreased with litter cover. We do not know how, and to what extent, individual species were responding to duff depth, but thicker duff layers occurred in projects with denser

vegetation which may have created cooler, adverse microsite conditions. As well, excessive duff may impede movement along the surface for some species, thereby decreasing mobility and lowering trap encounters (Greenslade 1964). With the range of species encountered, it is most probable that certain species were increasing under thick duff conditions while others were decreasing but it would seem that overall, thicker litter layers were not conducive to greater beetle abundance and biomass.

Soil characteristics, such as moisture, type, and pH have also been shown to affect beetle distribution (Holland and Luff 2000). Our data suggests that a CSM threshold existed for the sampled communities where excessive soil moisture resulted in lower abundance and biomass. While soil moisture is a function of precipitation, higher values can also be achieved with dense vegetative cover and/or thicker litter layers. Less than optimal conditions may lead to reduced prey availability or consumption which in turn may lead to lower fecundity or smaller individuals (Price 1984; Lövei and Sunderland 1996). The combination of increased duff depth and conserved soil moisture appeared to result in fewer beetles and reduced biomass. But because beetle communities tend to be in a constant state of flux, individual species responses may have differed given microsite and environmental conditions (Lövei and Sunderland 1996). Under one set of conditions certain species will proliferate while others perish. Communities were likely very different between projects as some species thrive in post-burn areas (Reed 1997; Larsen and Williams 1999), some in established stands (Lövei and Sunderland 1996), yet others are primarily edge species (Magura et al. 2001). Projects also had a high degree of heterogeneity and as such communities were likely very dynamic within a project, especially since vegetation, field perimeter, and wetland complexes were highly variable.

Carabid abundance appeared to be greatest in the second and third YPM although conserved soil moisture and vegetation density were the more important variables. Some studies have found Carabids to be more abundant the summer following a burn (Reed 1997; Larsen and Williams 1999) while others have found the opposite (Rickard 1974). Because we failed to capture the second peak of beetle activity (August – September), we can not be certain if the low numbers remained through the whole beetle active period of the first YPM. However, Larsen and Williams (1999) attributed a high degree of species richness post-burn, although not apparent until August, to the recolonization process with more diverse assemblages occurring several years after a fire. De Goede and van Dijk (1998) also found diversity increased yearly after a cultivated area was restored to grassland. While sampled projects had abundant beetles post-management and most likely had stable beetle communities pre-management, we can not be certain of the effects of a management event because communities were not sampled prior to disturbance. Lower Carabid abundance in the first growing season compared to subsequent seasons, does suggest negative impacts occurred with vegetation removal, regardless of removal approach.

With the exception of deer mice, the best mammal models did not include treatment or YPM parameters. While there were noticeable differences between treatment regimes for voles, models with vegetation characteristics, primarily density, provided a more parsimonious fit to the data. Shrews were not affected by treatment and YPM, but did appear to be dependent on vegetation density and forb composition. All relationships with visual obstruction readings were positive, indicating denser vegetation provided for greater abundance within these projects. Deer mice were most abundant the

first two YPM following vegetation removal, regardless of regime, but the relationship with vegetation density was also positive. Habitats with greater visual obstruction probably provide more security from predation plus more preferred microsite conditions for voles and shrews.

Shrew abundance appeared consistently in the best models for the vole abundance. There is evidence that northern short-tailed shrew (*Blarina brevicauda*) predation on vole nests may reduce vole populations (Barbehenn 1958; George et al. 1986) but very few *Blarina* were caught and all relationships between voles and shrews were positive. This suggests that they were most likely sharing habitats based on preference for similar site conditions. Although studies have found meadow voles in multiple habitats, their affinity for moist habitats with dense cover and high amounts of dead material is well documented (Geier and Best 1980; Yahner 1982; Klatt and Getz 1987; Doyle 1990; Bowles and Copsey 1992). The combination of vegetation density and duff depth appeared to be the strongest predictor of vole abundance. Including treatment and YPM variables produced inferior models although differences in vole abundance among treatment and YPM categories were noted. It follows that if voles prefer denser vegetation, more individuals should be captured in Tame-Hay and Native-Hay projects compared to Native-Burn projects which exhibited considerably lower measures of vegetation density and forb cover. Mean abundance in Native-Burn projects was indeed half that of other projects. Similar to Skinner et al. (1995), who found vole numbers peaked in the third year following conversion from cropland to dense nesting cover in the Aspen Parkland of Alberta, abundance within our study area was double in

YPM 2 through 4. Vegetation density and vole abundance were lowest in the first YPM with both reaching peak measures in the next two YPM.

Duff depth, in association with vegetation density, was also an important factor in vole response. Voles depend on the litter layer for construction of runways and nests, protection from predation, insulation from winter extremes, and forage in early spring when standing forage biomass is minimal (Peles and Barrett 1996). Snyder and Best (1988) found a positive relationship between meadow vole abundance and amount of dead plant material within 10 cm of the ground. Our study supports this finding in that abundance was lowest in areas with sparse vegetation and little duff. While burning and haying both remove standing dead material, it would appear that although the amount of stubble left post-haying may be adequate (greater than 10 cm), removal of cover by any means creates less than optimal meadow vole habitat during the first post-removal season (Peles and Barrett 1996). Voles also appeared to decrease somewhat as stands aged. This decrease may be a function of stand senescence (Naugle et al. 2000) and density-dependent feedbacks where the end result is not enough food ole densities (Turchin and Ostfeld 1997).

While native seeded stands, primarily a native-burn mix, often include native forbs, they may not provide legumes for which meadow voles have an affinity (Lindroth and Batzli 1984). Alfalfa, an abundant legume in tame mixes, is often incorporated into fields destined to be hayed. Hall et al. (1991) found vole densities were altered by changes in plant community structure, habitat cover, plant diversity, and food quality rather than food quantity. The presence of alfalfa as a desirable food source in combination with the higher vegetation density and duff may be primary in determining

meadow vole abundance (Basquill and Bondrup-Nielsen 1999*b*). Because seed mixes were highly variable among projects, density measures were not distinct between treatment regimes and this variability could be part of the failure to be able to predict vole responses to management.

Although deer mice are more of a habitat generalist than many small mammals, abundance has been shown to increase after disturbances such as fire or cultivation (Tester 1965; Snyder and Best 1988; Kaufman et al. 1983; Peterson et al. 1985; Skinner et al. 1995). This study supports the findings that deer mice tend to be colonizers, preferring open areas often in association with exposed soils and little litter cover (Tester 1965; Lysne 1991; Kaufman et al. 1988; Kaufman et al. 1990). While it was evident abundance was greatest in the first two YPM, other variables such as vegetation density, vole abundance, and Carabid abundance appeared to further explain the variation. Increased abundance with vegetation density may be a function of survivorship and protection from predation much like it is for voles. Although the relationship between deer mice and voles was weak, mice appeared to decrease as vole abundance increased. We did not directly measure competition among species but habitat utilization by deer mice is often determined through the inhibitory effect of other species, primarily meadow voles (Hallett et al. 1983). It has been shown that deer mice and voles can occupy the same habitat at low densities but as density increases, greater separation occurs such that mice are restricted to habitats not occupied by voles (Hallett et al. 1983). Therefore, it is possible deer mice were present in low densities with a management event creating less favorable conditions for voles but favorable conditions for deer mice, allowing for an irruption. Bowman et al. (2000) found deer mice responded to large-scale processes but

population growth occurred over a small spatial extent (< 300 m). While large-scale spatial irruptions (15 km) do occur (Steen et al. 1996), fine-scale (patch) responses can vary independently, depending on factors (e.g. resources) which tend to fluctuate in time and space (Krohne and Burgin 1990).

The use of Museum Special snap-traps often precludes the ability for shrew abundance analysis. These snap-traps are large, in comparison to a 3 g masked shrew, and as such are not known to return many shrew captures. The fact that shrews made up more than 15% of all captures presents an opportunity to address shrew use of planted cover. Since shrews are insectivorous, it was hypothesized that insect parameters would be primary explanatory variables. Shrews have been shown to persist in areas over long time frames (Getz and Hofmann 1999) but increase when food is abundant. *Blarina brevicauda* abundance increased more than four times the average density during a year of cicada emergence, providing a superabundance of food (Krohne et al. 1991). Although our best models included beetle abundance parameters, year was the single best explanatory variable with beetle abundance having similar average variable weights as the remaining covariates. This indicates that shrew abundance may be highly variable and because there were no strong relationships with any one predictor, that shrews were possibly responding inconsistently to vegetation density and forb cover, beetle abundance, or the combination thereof. Shrews are known to prefer moist areas with greater litter depth (Snyder and Best 1988). Although our results do not indicate duff depth or conserved soil moisture as primary explanatory factors, abundance did increase slightly with vegetation density, which can promote the formation of moist habitats. While beetle abundance was present in competing models, the relationship was negative.

Shrews ingest a wide variety of insect prey items and so may not be specifically seeking beetles (Martin et al. 1961). Tolerance of a wide range of habitat conditions may partially explain the lack of a single important explanatory variable other than annual variation.

Although there was a large decrease in the number of captures between 2000 and 2001, year was not a factor in vole and deer mice models suggesting that response to microsite conditions captured the annual variation. While there was a decrease of more than 50% in captures between sampling years, nine sites (Bigoraj, Gyorfi, Lambert, McLeod, Miller, Peterson, Start, Turner, and White) returned higher vole relative abundance in 2001. As with our study, Pasitschniak-Arts and Messier (1998), sampling in similar habitats and geographic locations, found relative abundance increased between sampling years in some areas but decreased in others, implying that population dynamics are highly variable and operating at a local scale. In a 25 year study of meadow vole population dynamics, Getz et al. (2001) failed to find evidence of cyclicity but rather, random fluctuations. Reasons for the decrease in captures between years remain unclear. Sampling was conducted in July and August which would have allowed for individuals from the first litter to be active. Some of the variation in relative abundance could be explained by the drought which was evident not only through decreased CSM values but also through vegetation characteristics such that the drier conditions and reduced growth may have impacted breeding success and survival. Also, the extent to which winter conditions affected survival was not measured, but winter die-off is normal in microtine life cycles and possibly aided in reducing abundance prior to the 2001 sampling session.

The extent to which removal sampling methods impacted abundance numbers is

uncertain but would have reduced the numbers to some extent. However, the sampling grid was small (0.16 ha) in comparison to the size of the projects (range from 6 ha to 62 ha) and should have allowed for re-population from the surrounding area. Andreassen and Ims (2001) found that higher dispersal rates in root voles (*Microtus oeconomus*) occurred most often with smaller patches (15 m X 15 m) and from areas of low densities into areas of even lower densities. They further suggest that patch specific conditions (patch size, demographic composition, spatiotemporal demographic variability) be incorporated when addressing dispersal rates. Microsite conditions were not taken into account and therefore, we cannot be certain about what caused the decline. It is possible that the reduction in capture numbers was a factor of a combination of the above conditions and a natural fluctuation. Because we witnessed a decrease in conserved soil moisture, reduced measures of vegetation parameters, and decreased small mammal abundance in association with each other, the consequence of drought conditions should not be ruled out. Emerging drought conditions were evident in 2000 such that continual dry conditions could have affected small mammal survival. Many wetlands that had held water the previous summer were dry at summers' end in 2001. Adverse changes to already dry habitat conditions, in association with unknown population parameters, may have caused the abundance dichotomy between 2000 and 2001

Small mammals as a group tended to be greater in Tame-Hay and Native-Hay stands with peak numbers in YPM 2 through 4 although management effects were not important parameters. This can be expected as voles made up the majority of captures and so their response was likely driving the model. Small mammals (voles, mice, and shrews) exhibited increased abundance primarily as a function of vegetation density. A

range of variables, including vegetation characteristics (Bowles and Copsey 1992; Snyder and Best 1988), patch connectivity (Basquill and Bondrup-Nielsen 1999*b*), along with study scale and area (Morris 1984; Dooley and Bowers 1996; Bowman et al. 2000) often determine small mammal abundance. However, spatial and temporal population dynamics tend to be largely a function of microhabitat conditions (Asher and Thomas 1984). While it is sufficient to say that year and vegetation density were better explanatory variables, there are intricacies within these systems and populations that are more than a function of these parameters. Vegetation conditions may be key in determining some small mammal populations, but other microsite and macrosite characteristics, which we did not explicitly test, may impact abundance. Research has implied a positive relationship between meadow voles and woodchuck (*Marmota monax*) burrows (Swihart and Picone 1995) or pocket gopher (*Thomomys spp.*) mounds (Whittaker et al. 1991). However, Klaas et al. (1998) suggested a negative relationship exists between meadow voles and pocket gopher mounds at small scales (300 m^2) but a positive one at larger scales (6,400 m^2). Temporal variation of habitat use has been shown for many small mammal species (Asher and Thomas 1984; Pasitschniak-Arts and Messier 1998) and Dooley and Bowers (1996) found meadow voles had relatively strong responses to microhabitat variation. Therefore, stand heterogeneity, particularly for larger tracts, may be important for allowing movements between microsites within the larger patch (Turchin and Ostfeld 1997).

As adjacent fields and habitats were not sampled for small mammals, we cannot be sure if our sites are unique within the landscape. Bird studies have suggested that permanent cover is highly preferred and in some situations acts as a source for certain

passerine species (McCoy et al. 1999; McMaster and Davis 2001). Pasitschniak-Arts and Messier (1998) determined that small mammal abundance differed between habitats in some years, but not all, and Basquill and Bondrup-Nielsen (1999*a*) found meadow vole density lower in habitats adjacent to meadows. But Harper et al. (1993) concluded patch sizes and shapes had no effect on population dynamics. It is possible the projects sampled were large enough to sustain most small mammal populations within themselves or, conversely, adjacent habitats were not impacting metapopulations. The role of adjacent habitats and patch size within this landscape needs to be addressed further. After initial conversion to nesting cover and burn events, small mammals must disperse from surrounding areas if not already present on site. Adjacent cropland may be harboring high densities of deer mice that spill out into newly managed cover while rights-of-way, fence rows, adjacent woodland, hayland, or pasture may be a source/refuge for many small mammals, indicating that at certain stages adjacent lands or pockets of different habitats are critical.

Three small mammal individuals could not be positively identified because only hair fragments were present on traps. Several specimens were in the process of being buried or consumed by Silphid beetles but identification was still possible. The extent with which specimens were removed from traps without observer knowledge is unknown although there was ample evidence to suggest this may be fairly common in areas of high Silphid density. Burying beetles (*Heterosilpha ramosa*) were commonly seen consuming bait from traps and Carrion beetles (*Necrophorus spp.*) were occasionally seen near specimens or empty traps. Small mammal studies should include an assessment of Silphid abundance given their ability to remove and bury carcasses within a few hours.

Without careful examination of each trap for hairs and the surrounding area, a carcass may be overlooked and as such would impact abundance calculations.

While it appears as though small mammal and beetle populations were not responding to management parameters specifically, they were responding to vegetation (microsite) characteristics impacted by management regimes. Overall, small mammal populations appeared to increase with vegetation density while beetles decreased with greater duff depth and conserved soil moisture. As pointed out earlier, there are relationships and factors within these systems which were not addressed (e.g. microsite conditions, spatial scale, habitat heterogeneity within the field and surrounding landscape, season long population changes, and pre-management populations). Although extensive areas may exhibit increased or decreased precipitation, population dynamics are very often random and localized. In periods of dry years, population responses and community dynamics may shift to reflect changes in microsite conditions. Agricultural benefits in the form of integrated pest control of crops may be enhanced through further understanding Carabid beetle response to established grasslands and the potential for permanent cover to act as source habitats. Conservation efforts of grassland species could be enhanced through knowledge of co-existing prey species since prey abundance or absence will affect predation pressure. Further long-term research is needed regarding relationships between micro- and macro-habitat, environmental conditions, and population responses given multiple management events – in essence prairie rodent and insect community ecology. Knowledge of how species react and interact in permanent cover under a range of conditions will allow for the appropriate management strategies given circumstances and goals of a situation.

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Table 2.1. Ranking of vegetation models relating year, management effects, and conserved soil moisture with visual obstruction readings, duff depth, maximum height, and forb composition. Models are ranked within each suite by Akaike's Information Criterion, adjusted for small sample size (ΔAIC_c), and model weight ratio (w/w_i). Competing models were selected as those with $\Delta AIC_c \leq 2$.

Suite and models	log (*L*)	K[a]	AIC_c	ΔAIC_c	w/w_i
VOR					
YR+TRT+YPM	71.1	14	105.56	0	1
YR+YPM	80.1	11	105.98	0.42	0.8103
YR+TRT+YPM+CSM	68.5	15	106.00	0.44	0.8031
YR	95.3	5	106.11	0.55	0.7598
Duff Depth					
TRT+CSM	101.6	7	117.16	0	1
TRT	104.7	6	117.85	0.70	0.7064
Maximum Height					
YR+YPM	185.0	11	210.88	0	1
Forb Composition					
TRT	594.7	6	607.85	0	1
YR+TRT	589.9	8	607.93	0.08	0.9620
YT+TRT+CSM	588.7	9	609.27	1.42	0.4915

Note: Model covariates are described in Appendix C.
[a] Number of model parameters including independent variables, dependent variable, intercept, random factor, and residual variance.

Table 2.2. Number of beetles caught, total counts per Family and year, and number of trap nights during pitfall trapping between May and July in 2000 and 2001 in the Aspen Parkland of Saskatchewan and Manitoba.

	Number Trapped		
Family	2000	2001	Total
Silphidae (Carrion beetle)	13,590	14,202	27,792
Carabidae (Ground beetle)	9,757	10,271	20,028
Curculionidae (Snout beetle)	1,281	665	1,946
Elateridae (Click beetle)	587	1,353	1,940
Cicindellidae (Tiger beetle)	353	442	795
Scarabidae (Dung beetle)	287	359	646
Tenebrionidae (Darkling beetle)	89	64	153
Histeridae (Hister beetle)	41	89	130
Byrrhidae (Pill beetle)	49	30	79
Dytiscidae (Predacious diving beetle)	15	7	22
Meliodae (Blister beetle)	8	13	21
Coccinellidae (Lady-bird beetle)	5	6	11
Hydrophilidae (Water scavenger beetle)	1	1	2
Total	26,063	27,502	53,565
Trap Nights (TN)	15,698	13,000	28,698

Table 2.3. Ranking of *a priori* management, *a priori* vegetation, landscape covariates, and exploratory models relating covariates with total beetle abundance. Models are ranked within each suite by Akaike's Information Criterion, adjusted for small sample size (ΔAIC_c), and model weight ratio (w/w_i). Competing models were selected as those with $\Delta AIC_c \leq 2$.

Suite and models	log (*L*)	K[a]	AIC_c	ΔAIC_c	w/w_i
A priori management suite					
YR	172.7	5	183.51	0	1
NULL	179.0	3	185.32	1.8	0.4056
A priori vegetation suite					
DUF+FORB	152.0	5	162.81	0	1
Landscape covariate suite					
$CSM+CSM^2$	169.6	5	180.41	0	1
CSM	172.7	4	181.23	0.82	0.6628
Exploratory suite					
$YR+DUF+FORB+CSM+CSM^2$	141.0	9	161.57	0	1
$DUF+FORB+CSM+CSM^2$	146.8	7	162.36	0.78	0.6757
DUF+FORB+CSM	149.4	6	162.55	0.98	0.6128
DUF+FORB	152.0	5	162.81	1.24	0.5381
YR+DUF+FORB	147.3	7	162.86	1.28	0.5260

Note: Model covariates are described in Appendix C.
[a] Number of model parameters including independent variables, dependent variable, intercept, random factor, and residual variance.

Table 2.4. Ranking of *a priori* management, *a priori* vegetation, landscape covariates, and exploratory models relating covariates with Carabid abundance. Models are ranked within each suite by Akaike's Information Criterion, adjusted for small sample size (ΔAIC_c), and model weight ratio (w/w_i). Competing models were selected as those with $\Delta AIC_c \leq 2$.

Suite and models	log (L)	K[a]	AIC_c	Δ AIC_c	w/w_i
A priori management suite					
NULL	169.3	3	175.62	0	1
YPM	156.8	9	177.37	1.76	0.4157
A priori vegetation suite					
DUF+DUF^2	160.2	5	171.01	0	1
DUF	162.6	4	171.13	0.12	0.9406
DUF+MXHT	161.4	5	172.21	1.20	0.5488
VOR+VOR^2	162.0	5	172.81	1.80	0.4066
DUF+FORB	162.3	5	173.11	2.10	0.3499
VOR+DUF	162.4	5	173.21	2.20	0.3329
Landscape covariate suite					
CSM+CSM^2	164.1	5	174.91	0	1
CROP+CROP^2	167.1	4	175.63	0.72	0.6968
CROP	167.2	4	175.73	0.82	0.6628
DNC+DNC^2	167.3	4	175.83	0.92	0.6305
WA+WA^2	168.1	4	176.63	1.72	0.4226
Exploratory suite					
YPM+VOR+VOR^2+CSM+CSM^2	133.5	13	165.02	0	1

Note: Model covariates are described in Appendix C.
[a] Number of model parameters including independent variables, dependent variable, intercept, random factor, and residual variance.

Table 2.5. Ranking of *a priori* management, *a priori* vegetation, landscape covariates, and exploratory models relating covariates with Carabid biomass. Models are ranked within each suite by Akaike's Information Criterion, adjusted for small sample size (ΔAIC_c), and model weight ratio (w/w_i). Competing models were selected as those with $\Delta AIC_c \leq 2$.

Suite and models	log (L)	K[a]	AIC_c	ΔAIC_c	w/w_i
A priori management suite					
YR	116.5	5	127.3	0	1
A priori vegetation suite					
VOR+FORB	109.9	5	120.71	0	1
DUF+MXHT	110.6	5	121.41	0.70	0.7047
DUF+FORB	111.0	5	121.81	1.10	0.5769
Landscape covariate suite					
CSM+CSM^2	119.9	5	130.71	0	1
CSM	123.6	4	132.13	1.42	0.4910
Exploratory suite					
YR+DUF+CSM+CSM^2	82.9	8	100.93	0	1
YR+VOR+DUF+CSM+CSM^2	81.8	8	102.37	1.44	0.2243

Note: Model covariates are described in Appendix C.
[a] Number of model parameters including independent variables, dependent variable, intercept, random factor, and residual variance.

Table 2.6. Number of small mammals caught, total counts per species and year, and number of trap nights operated in July of 2000 and 2001 in the Aspen Parkland of Saskatchewan and Manitoba.

Species	Number Trapped		
	2000	2001	Total
Meadow vole *(Microtus pennsylvanicus)*	633	298	921
Red-backed vole *(Clethrionomys gapperi)*	10	5	15
Deer mouse *(Peromyscus maniculatus)*	155	59	209
Masked shrew / Hayden's shrew *(Sorex cinereus/ haydeni)*	160	27	187
Arctic shrew *(Sorex arcticus)*	32	5	38
Northern short-tailed shrew *(Blarina brevicauda)*	9	3	12
13-lined ground squirrel *(Spermophilus tridecemlineatus)*	26	35	61
Northern pocket gopher *(Thomomys talpoides)*	8	20	28
Jumping mouse *(Zapus spp.)*	3	8	11
Olive-backed pocket mouse *(Perognathus fasciatus)*	1	0	1
unknown	1	2	3
Total	1038	462	1500
Trap Nights (TN)	10,000	10,250	20,250

Table 2.7. Ranking of *a priori* management, *a priori* vegetation, landscape covariates, and exploratory models relating covariates with vole abundance. Models are ranked within each suite by Akaike's Information Criterion, adjusted for small sample size (ΔAIC_c), and model weight ratio (w/w_i). Competing models were selected as those with $\Delta AIC_c \leq 2$.

Suite and models	log (L)	K[a]	AIC_c	ΔAIC_c	w/w_i
A priori management suite					
YR	196.2	5	207.01	0	1
TRT+YR	190.5	8	208.53	1.52	0.4683
A priori vegetation suite					
VOR+DUFF	190.5	5	201.31	0	1
VOR+MXHT	191.5	5	202.31	1.00	0.6065
VOR	194.2	4	202.73	1.42	0.4910
Landscape covariate suite					
SHRW	192.8	4	201.33	0	1
SHRW+SHRW^2	192.5	5	203.31	1.98	0.3720
Exploratory suite					
VOR+DUF+SHRW	181.1	6	194.25	0	1
VOR+MXHT+SHRW	181.1	6	194.25	0	1
VOR+SHRW	184.3	5	195.11	0.86	0.6504
VOR+FORB+SHRW	182.4	6	195.55	1.30	0.5220

Note: Model covariates are described in Appendix C.
[a] Number of model parameters including independent variables, dependent variable, intercept, random factor, and residual variance.

Table 2.8. Ranking of *a priori* management, *a priori* vegetation, landscape covariates, and exploratory models relating covariates with deer mouse abundance. Models are ranked within each suite by Akaike's Information Criterion, adjusted for small sample size (ΔAIC_c), and model weight ratio (w/w_i). Competing models were selected as those with $\Delta AIC_c \leq 2$.

Suite and models	log (*L*)	K[a]	AIC_c	ΔAIC_c	w/w_i
A priori management suite					
YPM	7.5	9	28.07	0	1
YR+YPM	3.8	11	29.68	1.61	0.4469
A priori vegetation suite					
VOR	28.1	4	36.63	0	1
VOR+FORB	27.5	5	38.31	1.68	0.4323
VOR+DUFF	27.8	5	38.61	1.98	0.3720
VOR+MXHT	27.9	5	38.71	2.08	0.3539
Landscape covariate suite					
DNC	32.4	4	40.93	0	1
NULL	35.1	3	41.42	0.48	0.7857
Exploratory suite					
YPM+VOR+VOLE	-1.0	11	24.88	0	1
YPM+VOR+CARB	-0.6	11	25.28	0.40	0.8187
YPM+CARB	-2.1	10	25.29	0.41	0.8163
YPM+VOR+VOLE+VOLE^2	-2.5	12	26.16	1.27	0.5288
YPM+MXHT+CARB	0.4	11	26.28	1.40	0.4966
YPM+CARB+CARB^2	0.5	11	26.38	1.50	0.4724
YPM+VOR+VOR^2+VOLE	-2.2	12	26.46	1.57	0.4551
YPM+VOR+DUF+VOLE	-2.0	12	26.66	1.77	0.4118
YPM+VOR	3.6	10	26.79	1.91	0.3856

Note: Model covariates are described in Appendix C.
[a] Number of model parameters including independent variables, dependent variable, intercept, random factor, and residual variance.

Table 2.9. Ranking of *a priori* management, *a priori* vegetation, landscape covariates, and exploratory models relating covariates with shrew abundance. Models are ranked within each suite by Akaike's Information Criterion, adjusted for small sample size (ΔAIC_c), and model weight ratio (w/w_i).

Suite and models	log (L)	K[a]	AIC_c	ΔAIC_c	w/w_i
A priori management suite					
YR	96.0	5	106.81	0	1
A priori vegetation suite					
VOR+FORB	128.0	5	138.81	0	1
Landscape covariate suite					
VOLE	130.3	4	138.83	0	1
VOLE+VOLE^2	129.9	5	140.71	1.88	0.3911
Exploratory suite					
YR+VOR+FORB+VOLE	87.2	8	105.23	0	1
YR+VOLE	92.7	6	105.85	0.62	0.7235
YR+VOR+FORB+VOLE+VOLE^2	85.7	9	106.27	1.04	0.5935
YR+VOR+FORB	90.9	7	106.46	1.23	0.5413
YR	96.0	5	106.81	1.58	0.4532

Note: Model covariates are described in Appendix C.
[a] Number of model parameters including independent variables, dependent variable, intercept, random factor, and residual variance.

Table 2.10. Ranking of *a priori* management, *a priori* vegetation, landscape covariates, and exploratory models relating covariates with total small mammal abundance. Models are ranked within each suite by Akaike's Information Criterion, adjusted for small sample size (ΔAIC_c), and model weight ratio (w/w_i).

Suite and models	log (L)	K[a]	AIC_c	ΔAIC_c	w/w_i
A priori management suite					
YR+TRT	152.5	8	170.53	0	1
YR	161.5	5	172.31	1.78	0.4102
A priori vegetation suite					
VOR	170.7	4	179.23	0	1
VOR+MXHT	169.1	5	179.91	0.68	0.7127
VOR+DUFF	169.6	5	180.41	1.18	0.5550
VOR+FORB	170.2	5	181.01	1.78	0.4112
Landscape covariate suite					
CARB+CARB^2	192.40	5	203.21	0	1
CSM+CSM^2	193.50	5	204.31	1.10	0.5769
BEET	195.90	4	204.43	1.22	0.5427
NULL	198.30	3	204.62	1.40	0.4954
Exploratory suite					
YR+VOR+FORB	148.2	7	163.76	0	1
YR+VOR	151.0	6	164.15	0.40	0.8207
YR+VOR+MXHT	148.6	7	164.16	0.40	0.8187
YR+VOR+CSM	148.9	7	164.46	0.70	0.7047
YR+VOR+MXHT+CROP	146.5	8	164.53	0.77	0.6796
YR+VOR+MXHT+CSM	146.5	8	164.53	0.77	0.6796
YR+VOR+FORB+CROP	146.5	8	164.53	0.77	0.6796
YR+VOR+DUF	149.0	7	164.56	0.80	0.6703
YR+VOR+CROP	149.4	7	164.96	1.20	0.5488
YR+VOR+DUF+CSM	147.0	8	165.03	1.27	0.5292
YR+VOR+DNC	149.8	7	165.36	1.60	0.4493
YR+TRT+VOR	144.9	9	165.47	1.72	0.4240
YR+VOR+FORB+DNC	147.5	8	165.53	1.77	0.4122
YR+VOR+FORB+BEET	147.5	8	165.53	1.77	0.4122
YR+VOR+FORB+CSM	147.6	8	165.63	1.87	0.3921
YR+VOR+DUF+CROP	147.7	8	165.73	1.97	0.3729

Note: Model covariates are described in Appendix C.

[a] Number of model parameters including independent variables, dependent variable, intercept, random factor, and residual variance.

Figure 2.1. Ecoregion boundaries and locations of study sites sampled during 2000 and 2001 in Saskatchewan and Manitoba. Study sites are symbol coded to treatment type.

Figure 2.2. Schematic diagram of study design showing advancement of projects in stand age post-management (YPM) between sampling years 2000 and 2001.

Figure 2.3. Diagram of a pitfall trap used for insect sampling and placement of the trap at ground level. The trap consists of an outer plastic cup and removable inner liner filled with a saturated salt solution.

Figure 2.4. Diagram of the 5 by 5 grid layout used in small mammal sampling. Two Museum Special snap-traps were placed at each grid point with 10 m spacing between grid points.

Figure 2.5. Mean visual obstruction (least square means ± 1SE) among treatments for projects sampled in Saskatchewan and Manitoba during 2000 and 2001.

Figure 2.6. Mean visual obstruction (least square means ± 1SE) per year post-management for projects sampled in Saskatchewan and Manitoba during 2000 and 2001.

Figure 2.7. Relationships between conserved soil moisture, CSM, and visual obstruction readings within each treatment regime for projects sampled in Saskatchewan and

Manitoba during 2000 and 2001. CSM, at May 1, is derived from the monthly precipitation records in the preceding 21 months.

Figure 2.8. Mean duff depth (least square means ± 1SE) per treatment type for projects sampled in Saskatchewan and Manitoba during 2000 and 2001.

Figure 2.9. Relationships between conserved soil moisture, CSM, and duff depth within each treatment regime for projects sampled in Saskatchewan and Manitoba during 2000 and 2001. CSM, at May 1, is derived from the monthly precipitation records in the preceding 21 months.

Figure 2.10. Mean maximum height (least square means ± 1SE) per year post-management for projects sampled in Saskatchewan and Manitoba during 2000 and 2001.

Figure 2.11. Mean percent forb composition ($[\sin x]^2$ of the least square means) ± 95% CI within treatment regimes sampled in Saskatchewan and Manitoba in 2000 and 2001.

Figure 2.12. Relationship between duff depth and beetle abundance ($\ln[x+1]$ of the number captured per 100 trap nights) ± 95% CI for projects sampled between May and July of 2000 and 2001 in Saskatchewan and Manitoba.

Figure 2.13. Relationship between forb composition (arcsine transformed) and total beetle abundance (ln[x+1] of number captured per 100 trap nights) ±95% CI for projects sampled between May and July of 2000 and 2001 in Saskatchewan and Manitoba.

Figure 2.14. Relationship between conserved soil moisture, CSM, and beetle abundance (ln[x+1] of the number captured per 100 trap nights) for projects sampled between May and July of 2000 and 2001 in Saskatchewan and Manitoba. CSM, at May 1, is derived from the monthly precipitation records for the preceding 21 months.

Figure 2.15. Relationship between duff depth and ground beetle (Carabidae) abundance (ln[x+1] of the number captured per 100 trap nights) for projects sampled between May and July of 2000 and 2001 in Saskatchewan and Manitoba.

Figure 2.16. Relationship between conserved soil moisture, CSM, and ground beetle (Carabidae) abundance (ln[x+1] of the number captured per 100 trap nights) for projects sampled May through July of 2000 and 2001in Saskatchewan and Manitoba. CSM, at May 1, is derived from the monthly precipitation records for the preceding 21 months.

Figure 2.17. Relationship between visual obstruction and ground beetle (Carabidae) abundance (ln[x+1] of the number captured per 100 trap nights) for projects sampled in Saskatchewan and Manitoba between May and July of 2000 and 2001.

Figure 2.18. Mean ground beetle (Carabidae) abundance (mean number captured per 100 trap nights) ±95% CI per growing season for projects sampled in Saskatchewan and Manitoba between May and July of 2000 and 2001.

Figure 2.19. Relationship between forb composition (arcsine transformed) and ground beetle (Carabidae) biomass (ln[x+1] of g/m^2) ±95% CI for projects sampled in Saskatchewan and Manitoba between May and July of 2000 and 2001.

Figure 2.20. Relationship between duff depth and ground beetle (Carabidae) biomass (ln[x+1] of g/m^2) ±95% CI for projects sampled in Saskatchewan and Manitoba between May and July of 2000 and 2001.

Figure 2.21. Relationship between conserved soil moisture, CSM, and ground beetle (Carabidae) biomass (ln[x+1] of g/m^2) for projects sampled in Saskatchewan and Manitoba between May and July of 2000 and 2001. CSM, at May 1, is derived from the monthly precipitation records for the preceding 21 months.

Figure 2.22. Relationship between visual obstruction and vole abundance (ln[x+1] of the number captured per 100 trap nights) ±95% CI for projects sampled in Saskatchewan and Manitoba in July of 2000 and 2001.

Figure 2.23. Relationship between duff depth and vole abundance (ln[x+1] of the number captured per 100 trap nights) ±95% CI for projects sampled in July of 2000 and 2001 in Saskatchewan and Manitoba.

Figure 2.24. Relationship between shrew abundance and vole abundance (both abundance estimates are ln[x+1] of number captured per 100 trap nights) ±95% CI for projects sampled in July of 2000 and 2001 in Saskatchewan and Manitoba.

Figure 2.25. Mean deer mice abundance ([x-1]/x of least square means of 1/[x+1] of number captured per 100 trap nights) ±95% CI per year post-management for projects sampled in July of 2000 and 2001 in Saskatchewan and Manitoba.

Figure 2.26. Relationship between visual obstruction and deer mice abundance (1/[x+1] of the number captured per 100 trap nights) ±95% CI for projects sampled in July of 2000 and 2001 in Saskatchewan and Manitoba.

Figure 2.27. Relationship between vole abundance (ln[x+1] of number captured per 100 trap nights) and deer mice abundance (1/[x+1] of number captured per 100 trap nights) ±95% CI for projects sampled in July of 2000 and 2001 in Saskatchewan and Manitoba.

Figure 2.28. Relationship between visual obstruction and shrew abundance (ln[x+1] of number captured per 100 trap nights) ±95% CI for projects sampled in July of 2000 and 2001 in Saskatchewan and Manitoba.

Figure 2.29. Relationship between forb composition (arcsine transformed) and shrew abundance (ln[x+1] of number captured per 100 trap nights) ±95% CI for projects sampled in July of 2000 and 2001 in Saskatchewan and Manitoba.

Figure 2.30. Mean abundance of voles, shrews, deer mice, and total small mammals per growing season for projects sampled in July 2000 and 2001 in Saskatchewan and Manitoba. abundances are presented as the antilog[x]-1 of least square means for vole, shrew, and total catch effort, and [x-1]/x of least square means for deer mice catch effort. Catch effort is calculated as the corrected number of individuals captured per 100 trap nights.

Figure 2.31. Mean small mammal abundance (antilog[x]-1 of the least square mean number captured per 100 trap nights) ±95% CI per growing season for projects sampled in July of 2000 and 2001 in Saskatchewan and Manitoba.

Figure 2.32. Relationship between visual obstruction and small mammal abundance (ln[x+1] of number captured per 100 trap nights) ±95% CI for projects sampled in July of 2000 and 2001in Saskatchewan and Manitoba.

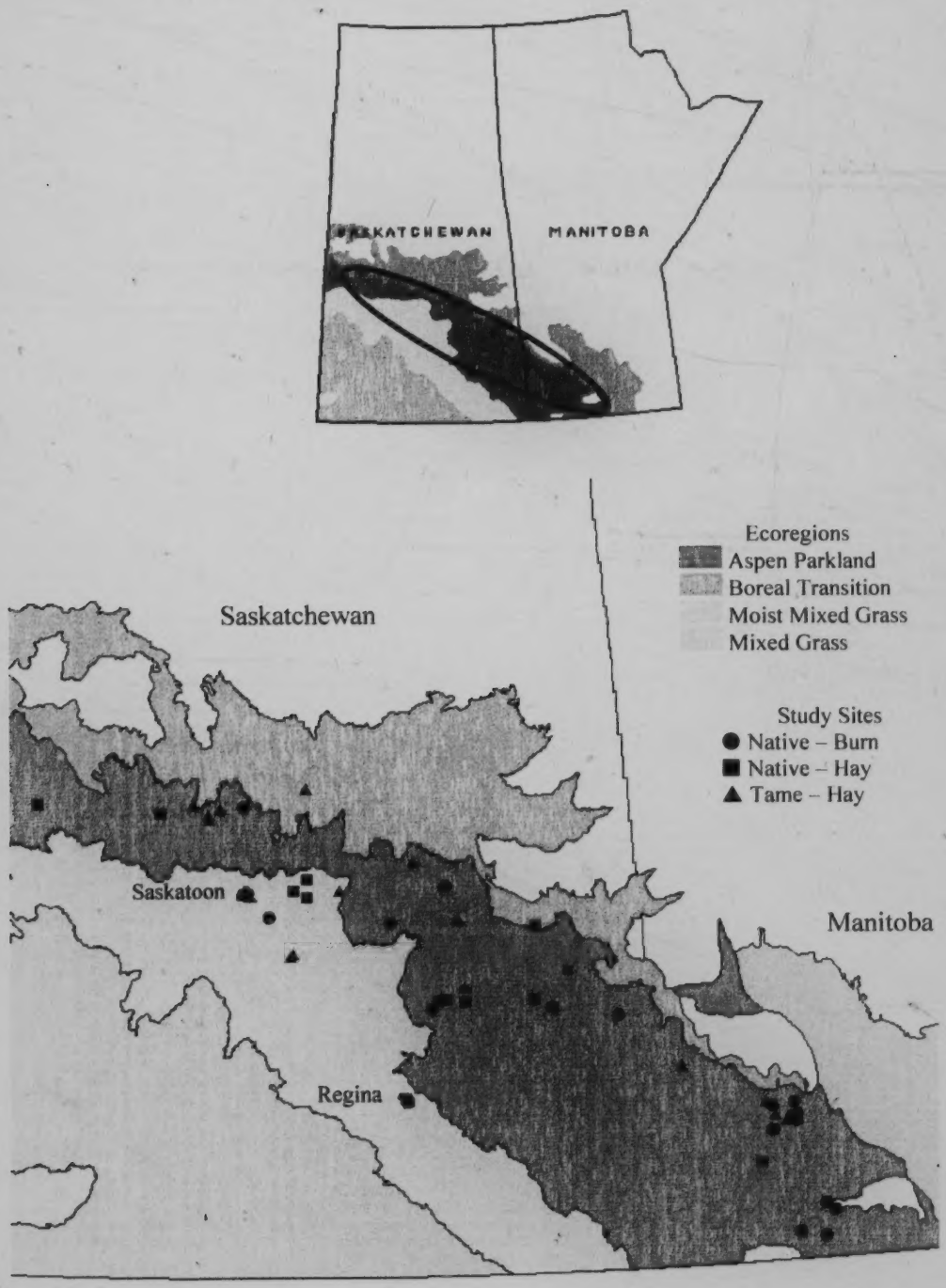
MANITOBA
Ecoregions
Aspen Parkland
Boreal Transition
Moist Mixed Grass
Mixed Grass
Study Sites
Native – Burn
Native – Hay
Tame – Hay
Saskatchewan
Saskatoon
Manitoba
Regina

Figure 2.1

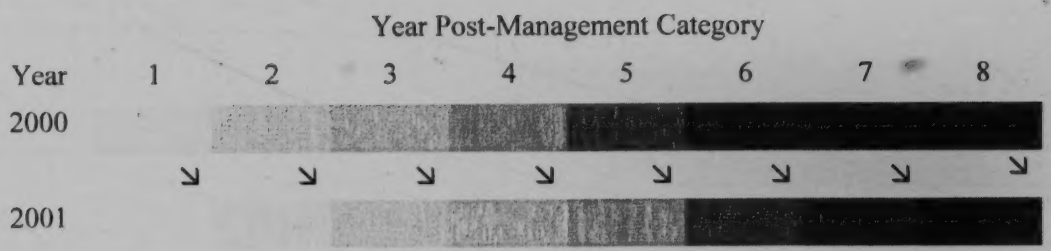

Figure 2.2

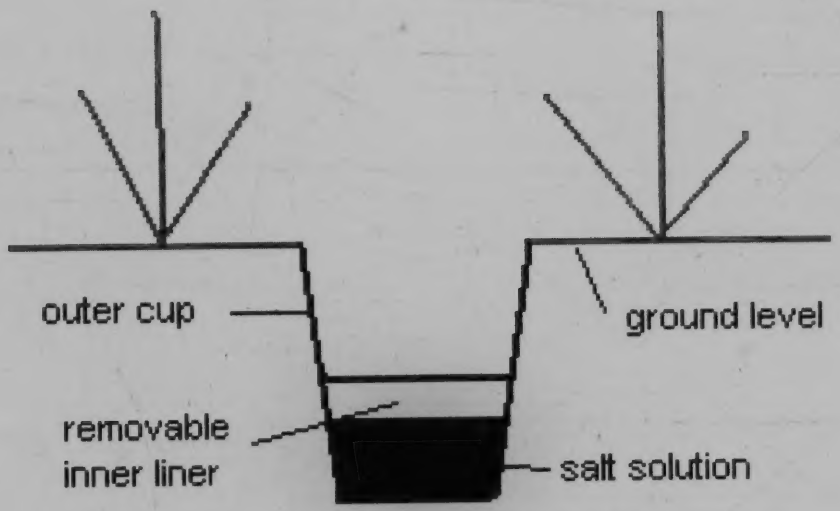
outer cup
ground level
removable
inner liner
salt solution

Figure 2.3

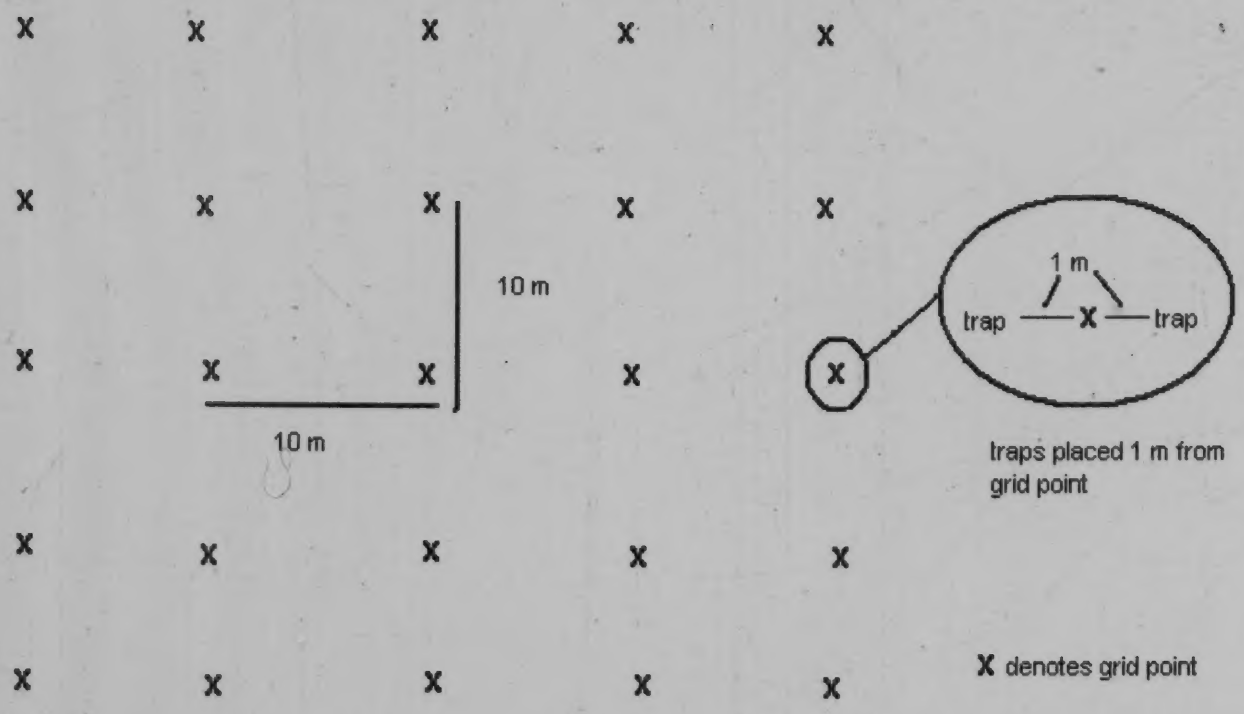
10 m
10 m
1 m
trap
trap
traps placed 1 m from grid point
X denotes grid point

Figure 2.4

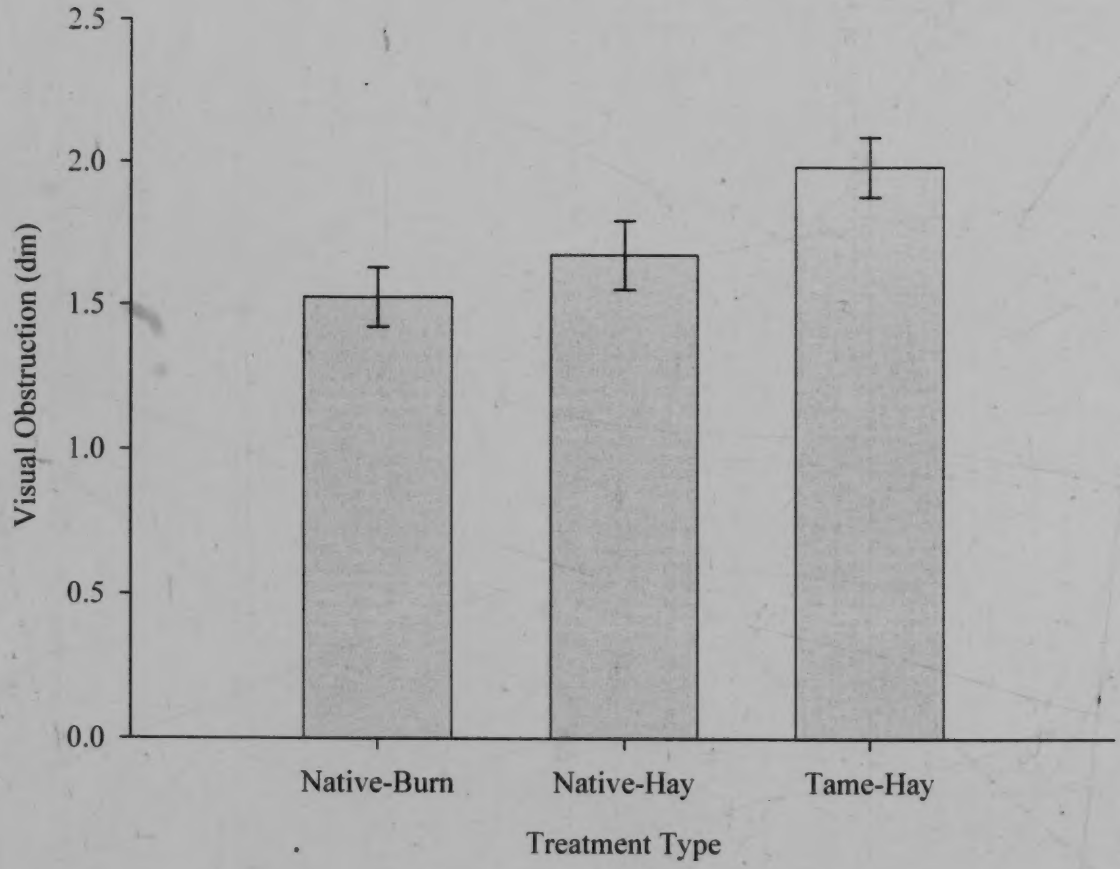
2.5
2.0
1.5
1.0
0.5
0.0
Visual Obstruction (dm)
Native-Burn
Native-Hay
Tame-Hay
Treatment Type

Figure 2.5

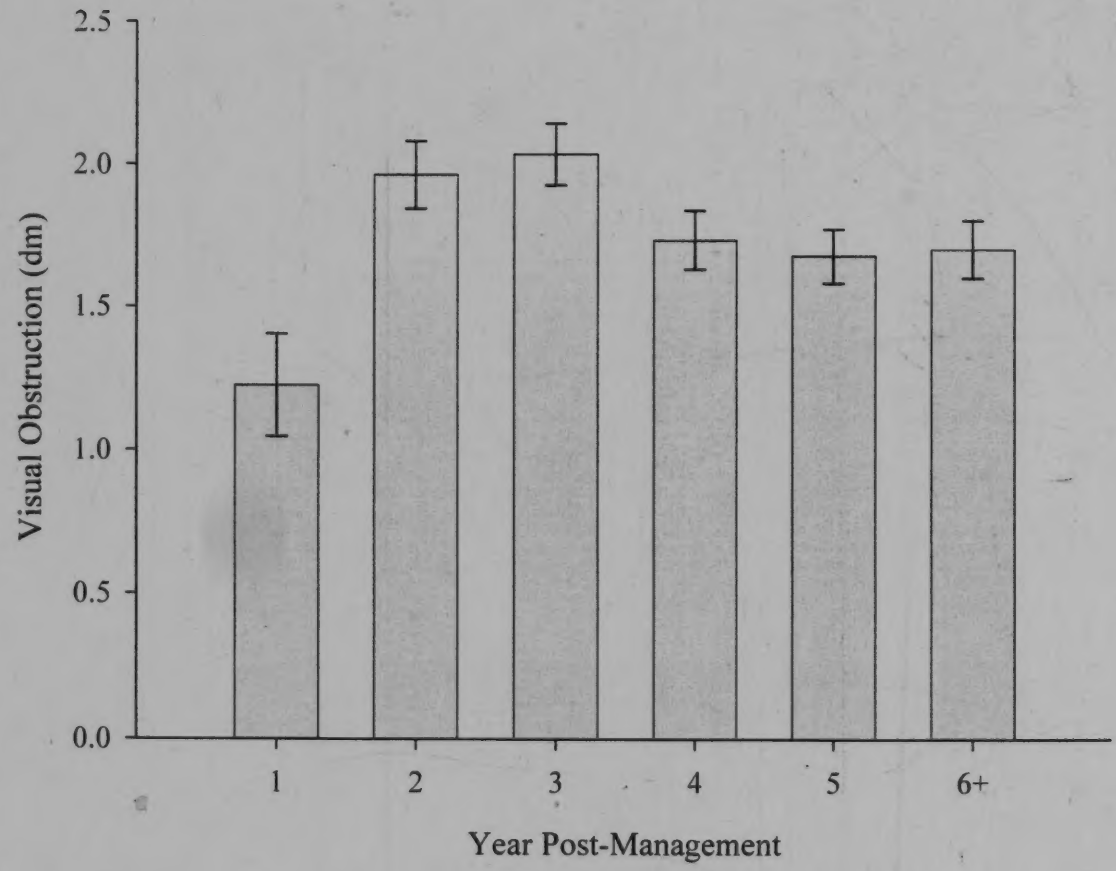
2.5
2.0
1.5
1.0
0.5
0.0
Visual Obstruction (dm)
1
2
3
4
5
6+
Year Post-Management

Figure 2.6

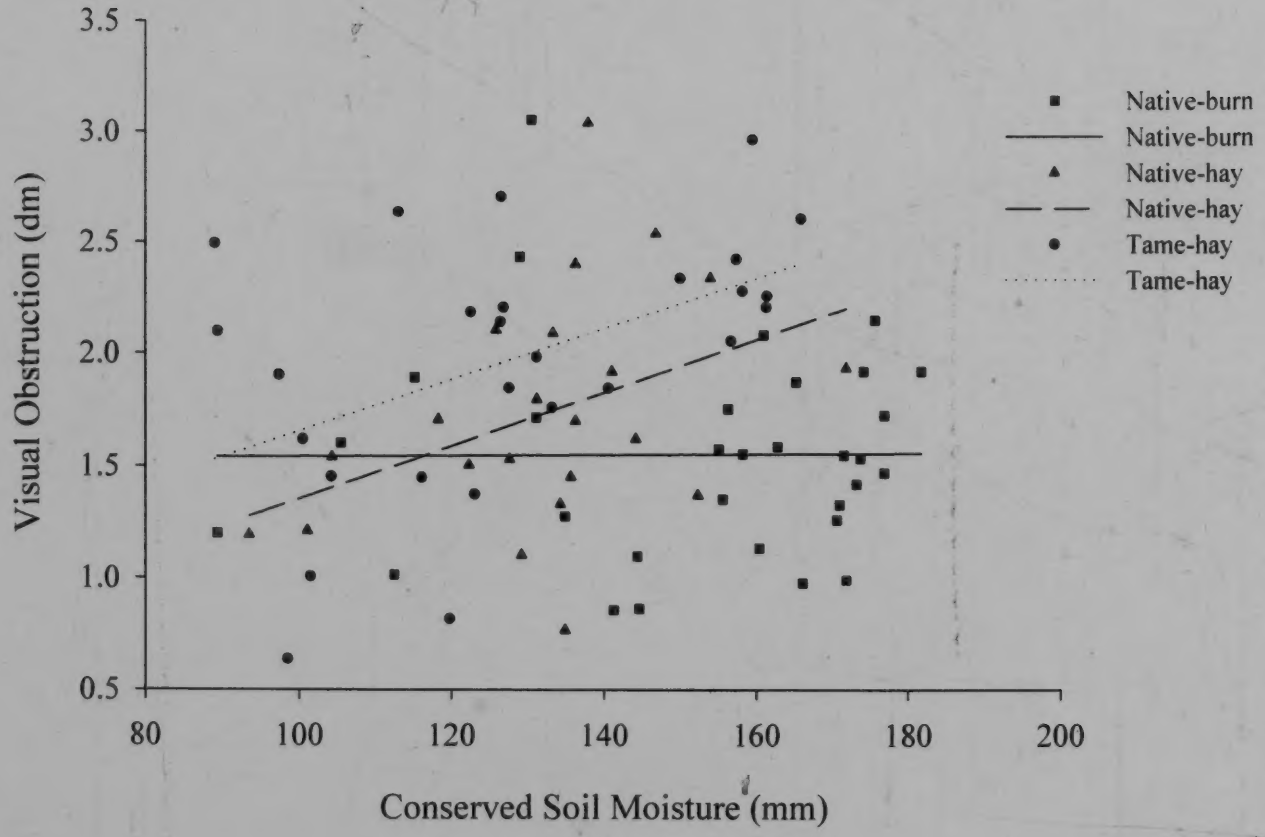
Visual Obstruction (dm)
3.5
3.0
2.5
2.0
1.5
1.0
0.5
80
100
120
140
160
180
200
Conserved Soil Moisture (mm)
Native-burn
Native-burn
Native-hay
Native-hay
Tame-hay
Tame-hay

Figure 2.7

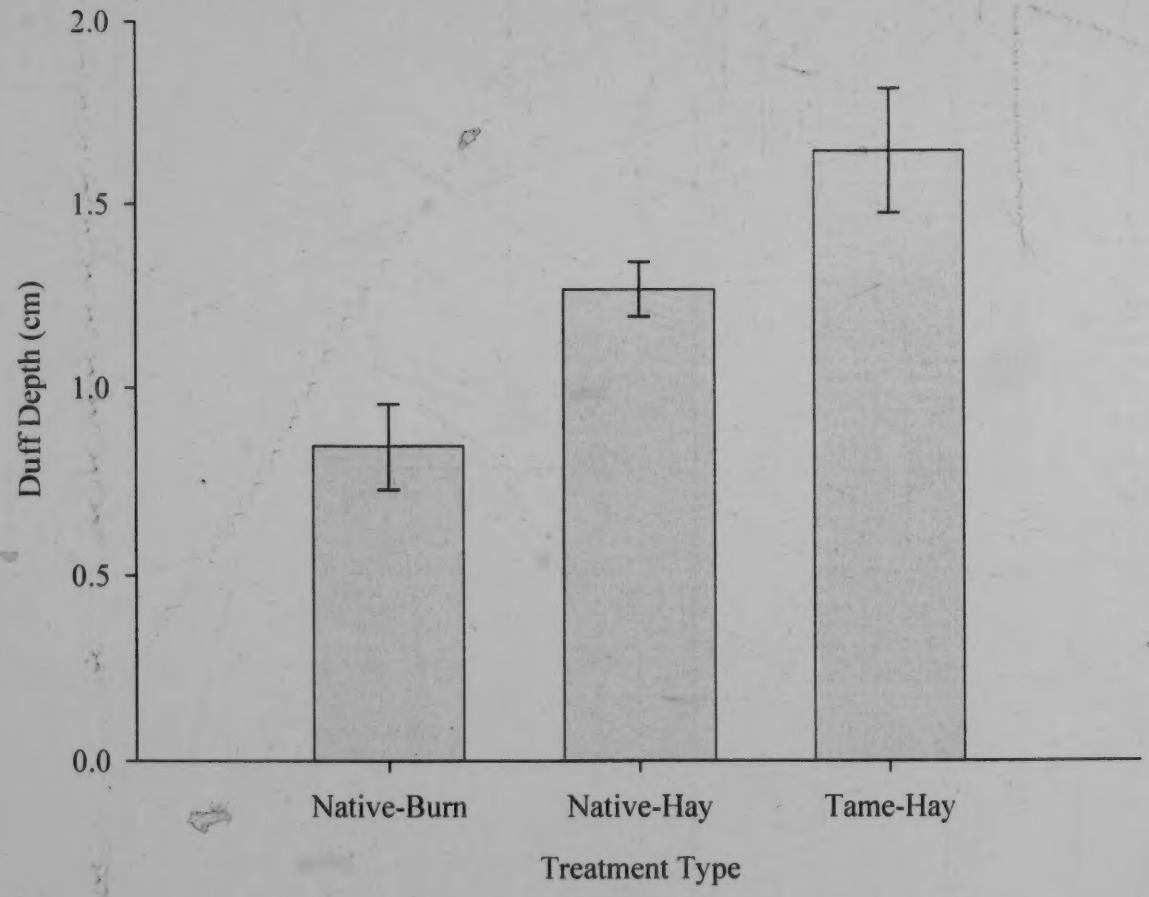
2.0
1.5
1.0
0.5
0.0
Duff Depth (cm)
Native-Burn
Native-Hay
Tame-Hay
Treatment Type

Figure 2.8

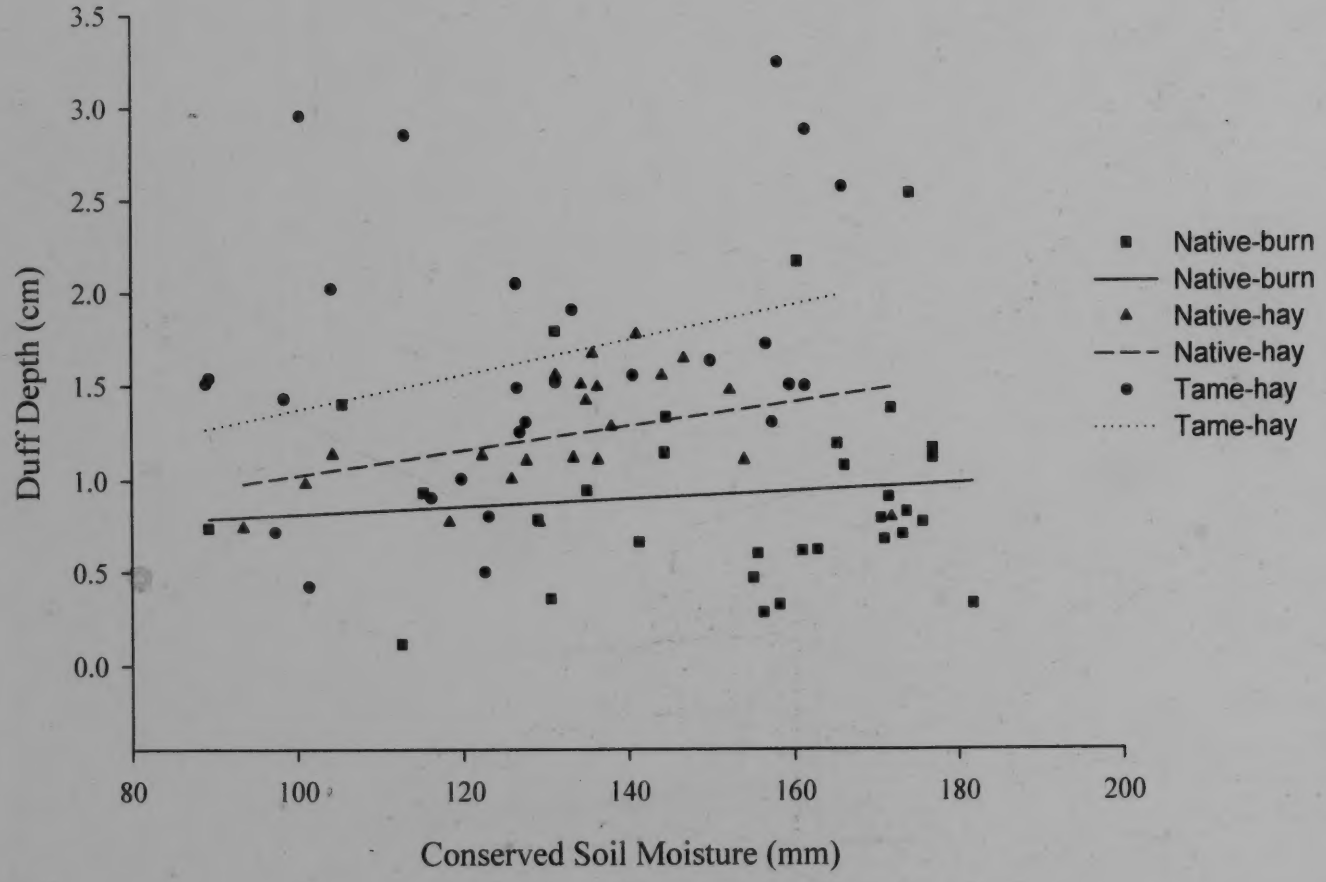
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
Duff Depth (cm)
80
100
120
140
160
180
200
Conserved Soil Moisture (mm)
Native-burn
Native-burn
Native-hay
Native-hay
Tame-hay
Tame-hay

Figure 2.9

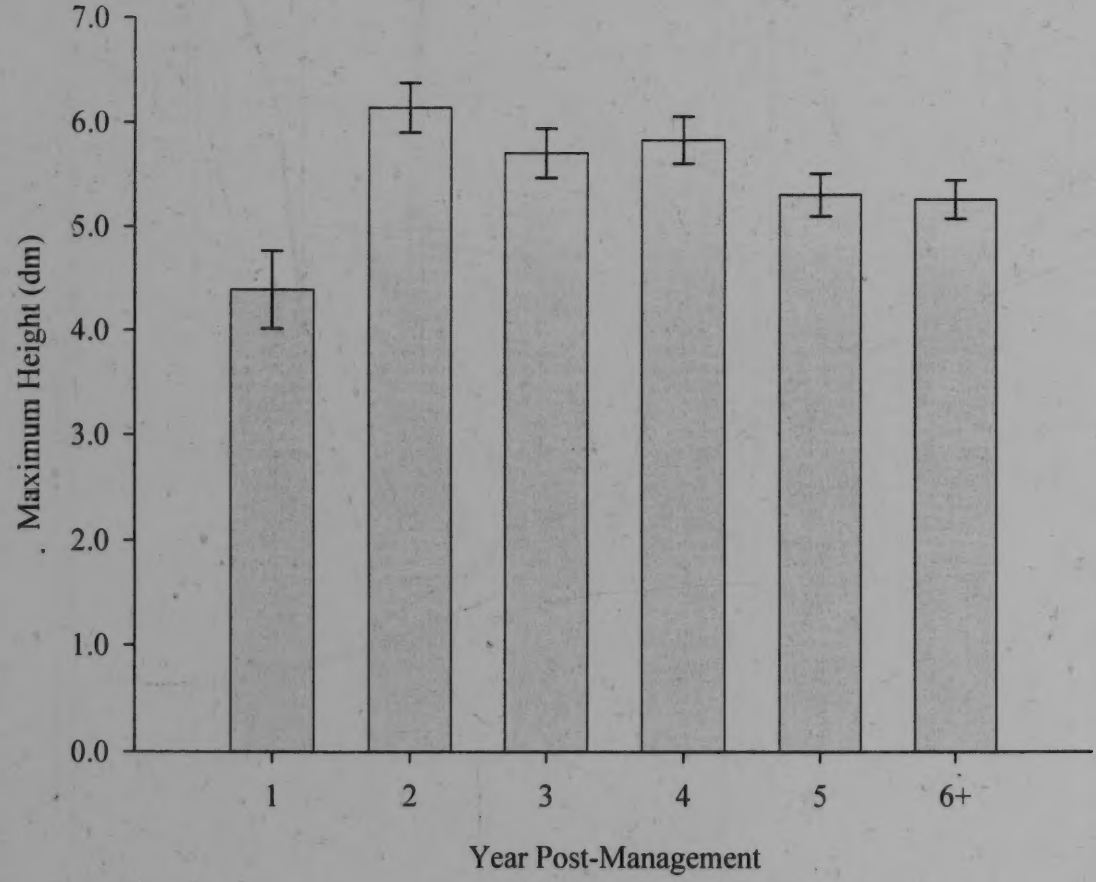
7.0
6.0
5.0
4.0
3.0
2.0
1.0
0.0
Maximum Height (dm)
1
2
3
4
5
6+
Year Post-Management

Figure 2.10

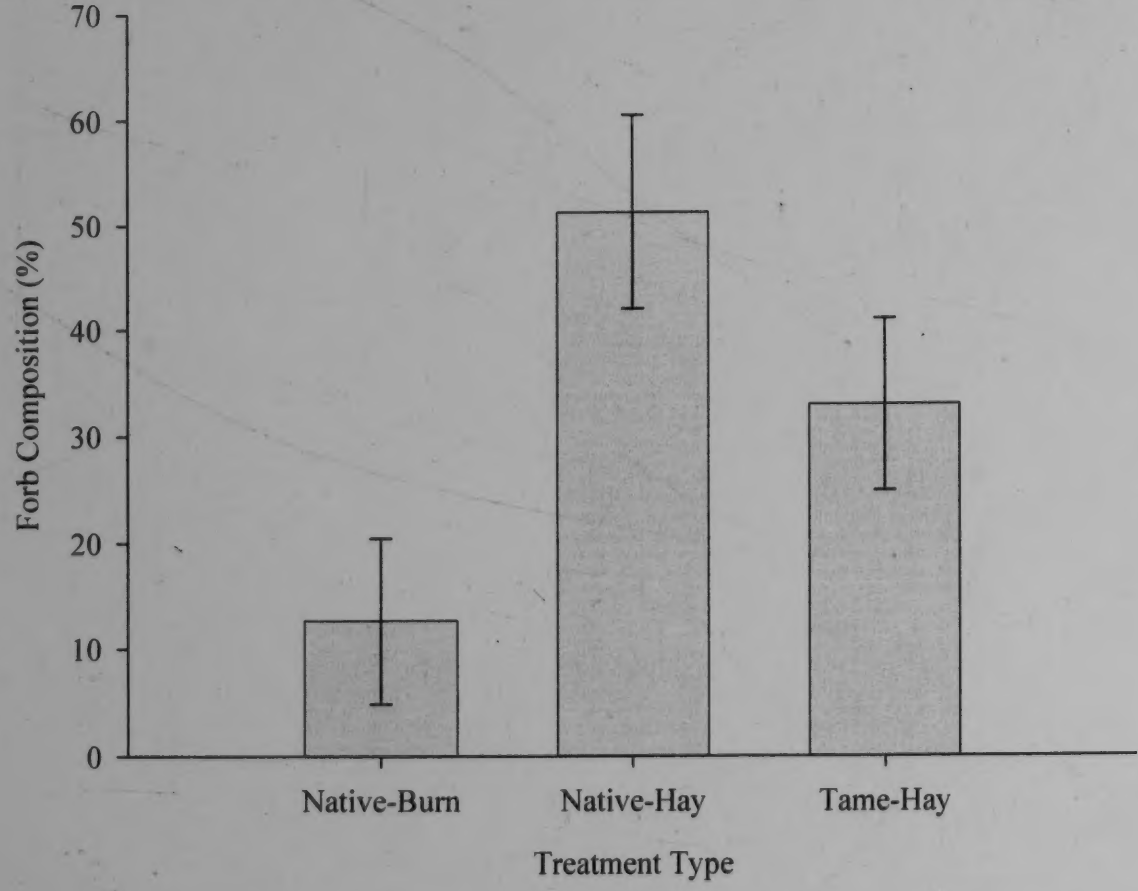
70
60
50
40
30
20
10
0
Forb Composition (%)
Native-Burn
Native-Hay
Tame-Hay
Treatment Type

Figure 2.11

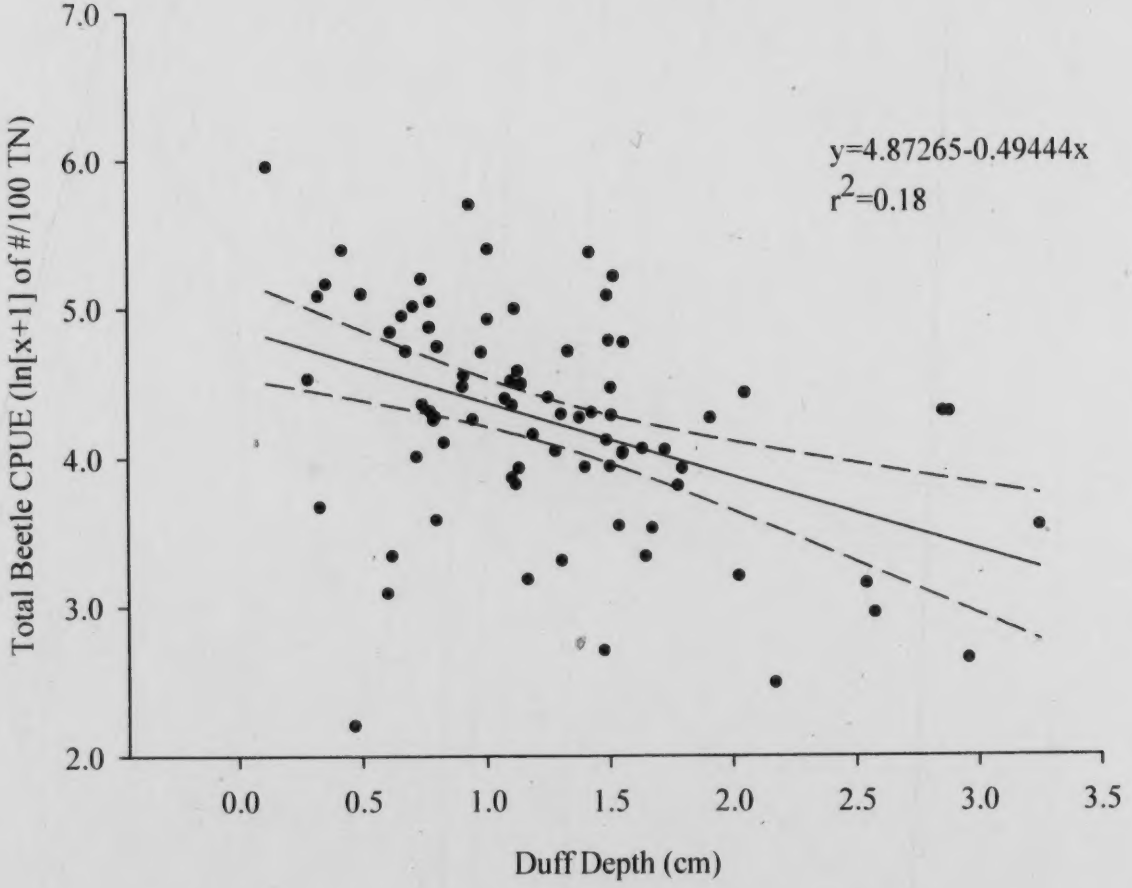
y=4.87265-0.49444x
r2=0.18
Total Beetle CPUE (ln[x+1] of #/100 TN)
Duff Depth (cm)
7.0
6.0
5.0
4.0
3.0
2.0
0.0
0.5
1.0
1.5
2.0
2.5
3.0
3.5

Figure 2.12

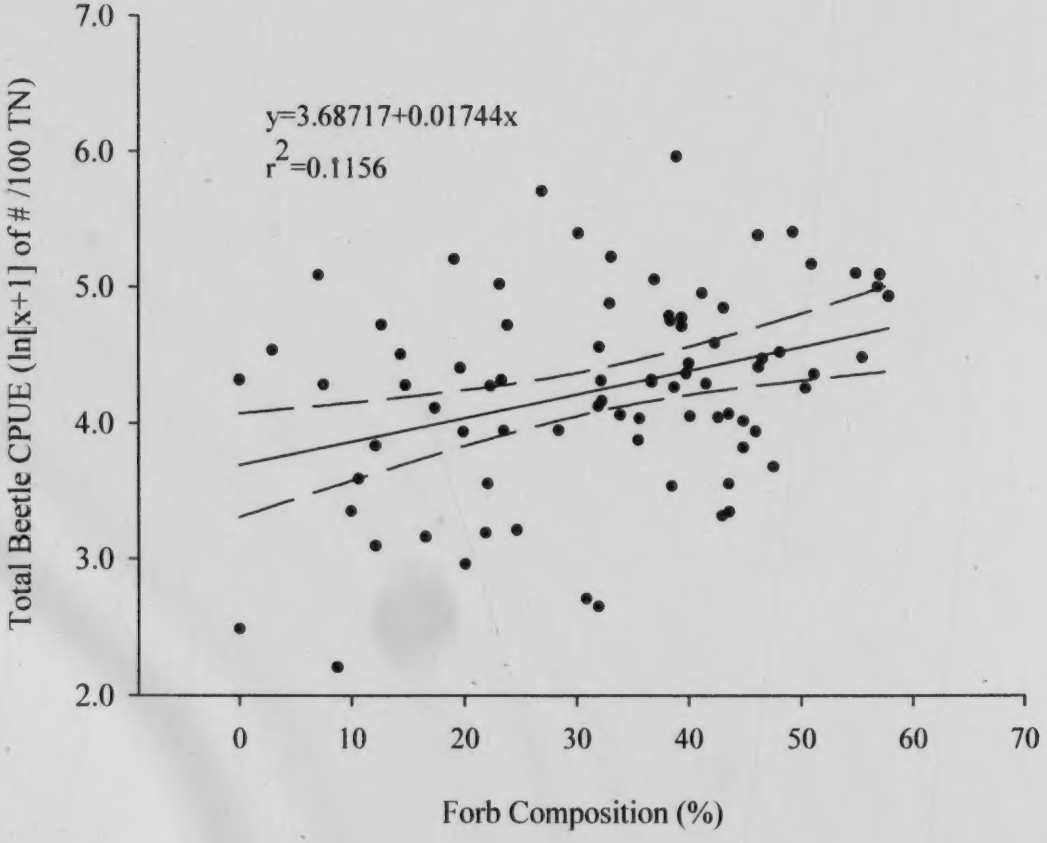
y=3.68717+0.01744x
r2=0.1156
Total Beetle CPUE (ln[x+1] of # /100 TN)
Forb Composition (%)
7.0
6.0
5.0
4.0
3.0
2.0
0
10
20
30
40
50
60
70

Figure 2.13

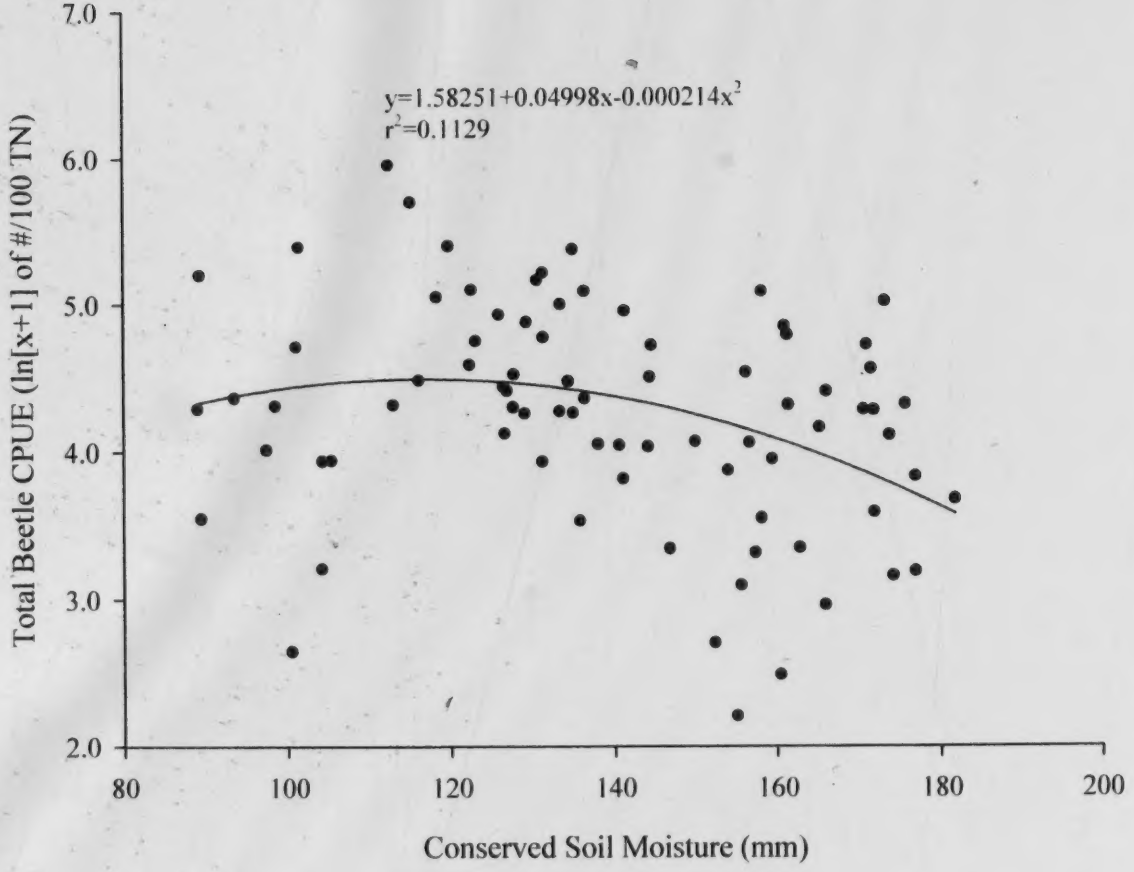
y=1.58251+0.04998x-0.000214x²
r²=0.1129
Total Beetle CPUE (ln[x+1] of #/100 TN)
7.0
6.0
5.0
4.0
3.0
2.0
80
100
120
140
160
180
200
Conserved Soil Moisture (mm)

Figure 2.14

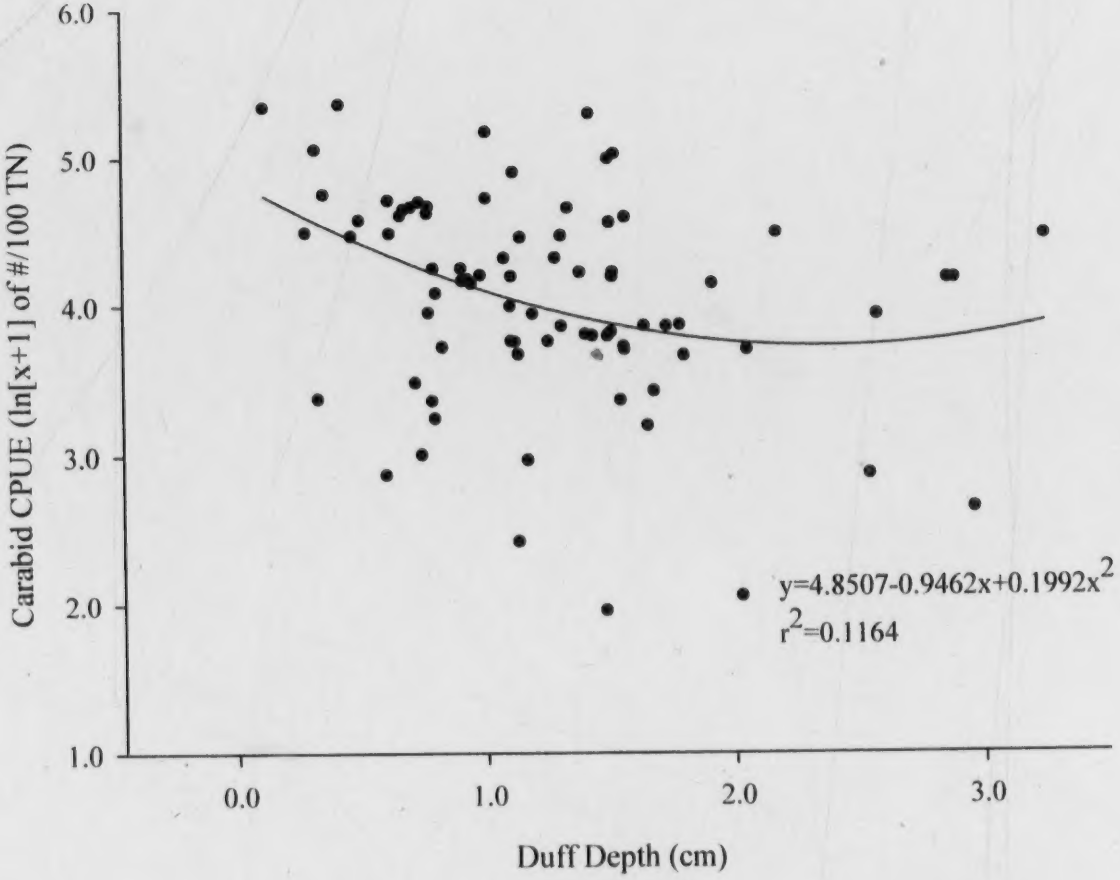
6.0
5.0
4.0
3.0
2.0
1.0
Carabid CPUE (ln[x+1] of #/100 TN)
0.0
1.0
2.0
3.0
Duff Depth (cm)
y=4.8507-0.9462x+0.1992x2
r2=0.1164

Figure 2.15

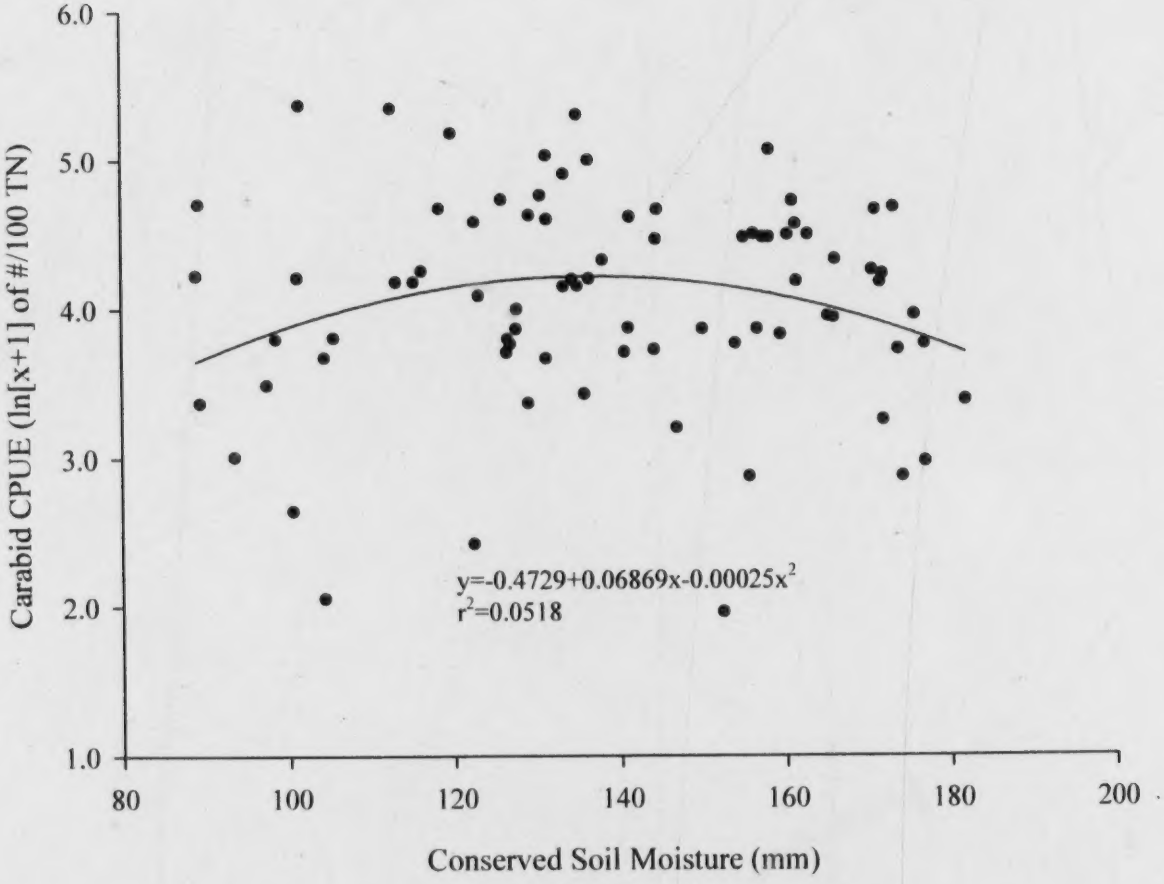
6.0
5.0
4.0
3.0
2.0
1.0
Carabid CPUE (ln[x+1] of #/100 TN)
y=-0.4729+0.06869x-0.00025x²
r²=0.0518
80
100
120
140
160
180
200
Conserved Soil Moisture (mm)

Figure 2.16

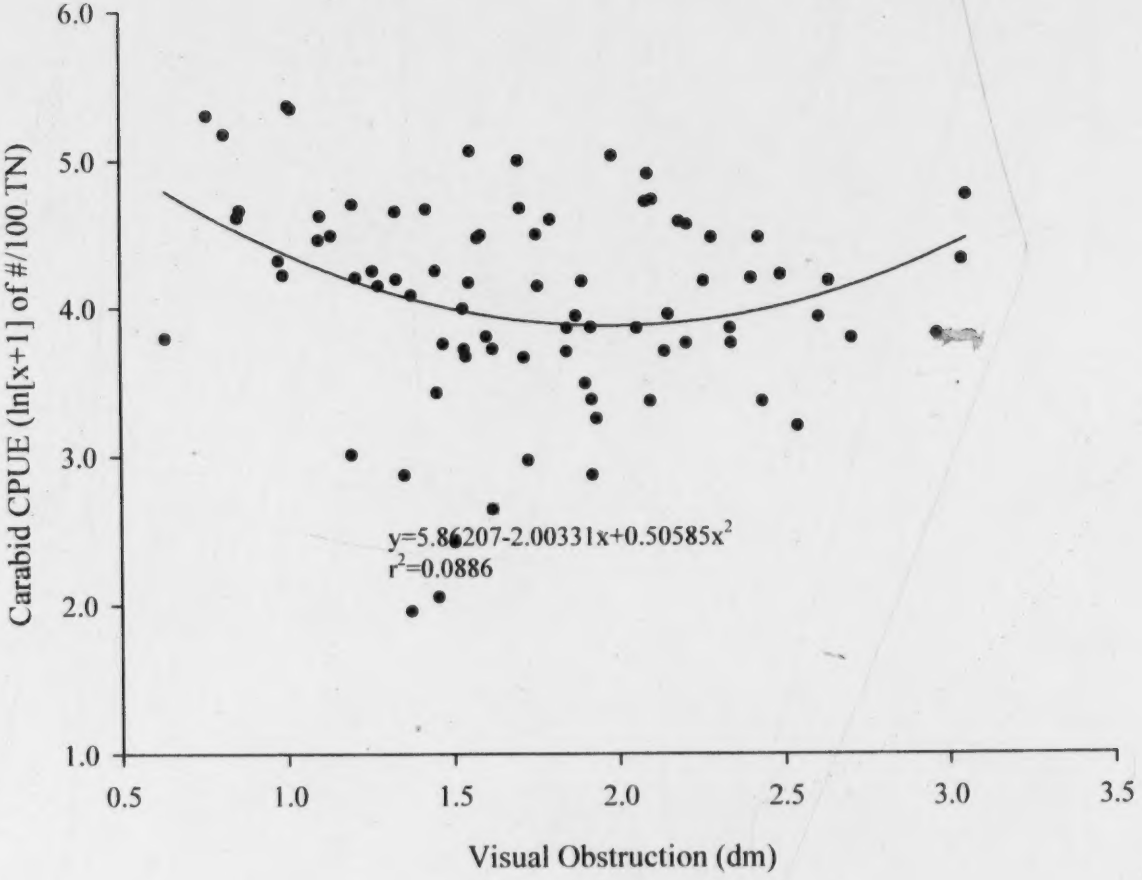
Carabid CPUE (ln[x+1] of #/100 TN)
6.0
5.0
4.0
3.0
2.0
1.0
0.5
1.0
1.5
2.0
2.5
3.0
3.5
Visual Obstruction (dm)
y=5.86207-2.00331x+0.50585x²
r²=0.0886

Figure 2.17

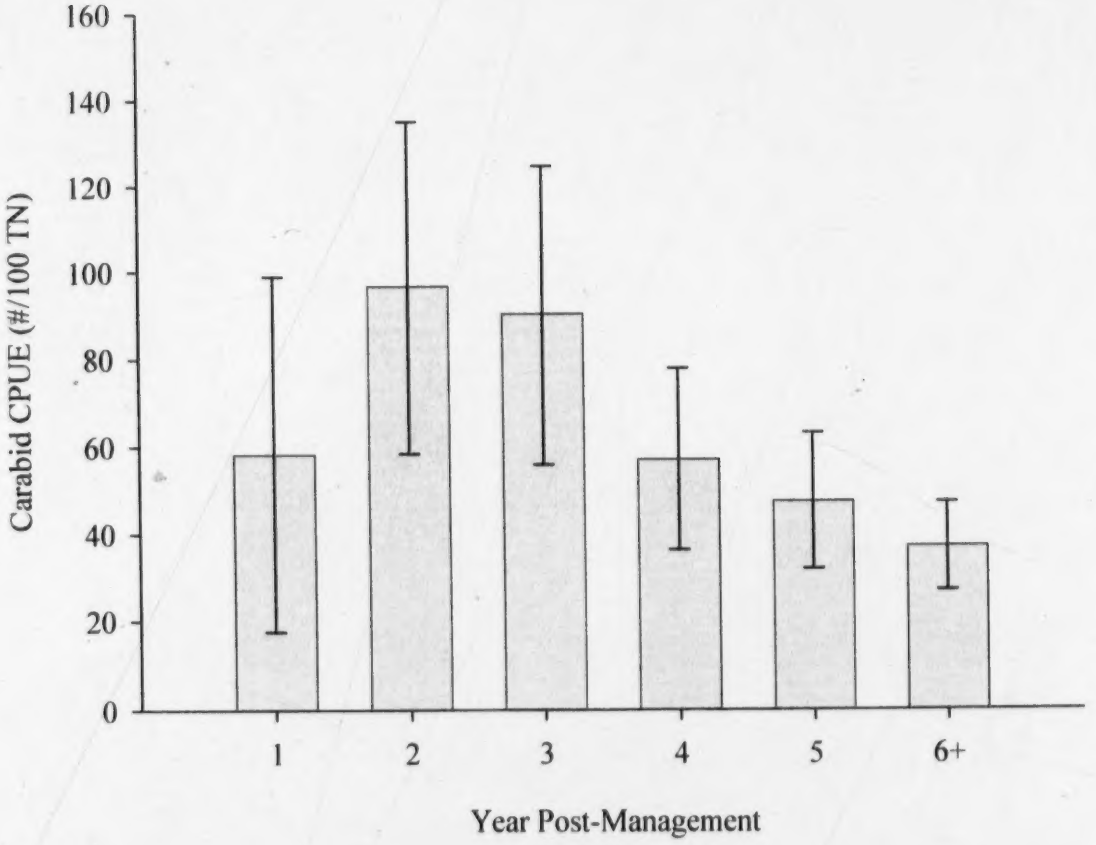
160
140
120
100
80
60
40
20
0
Carabid CPUE (#/100 TN)
1
2
3
4
5
6+
Year Post-Management

Figure 2.18

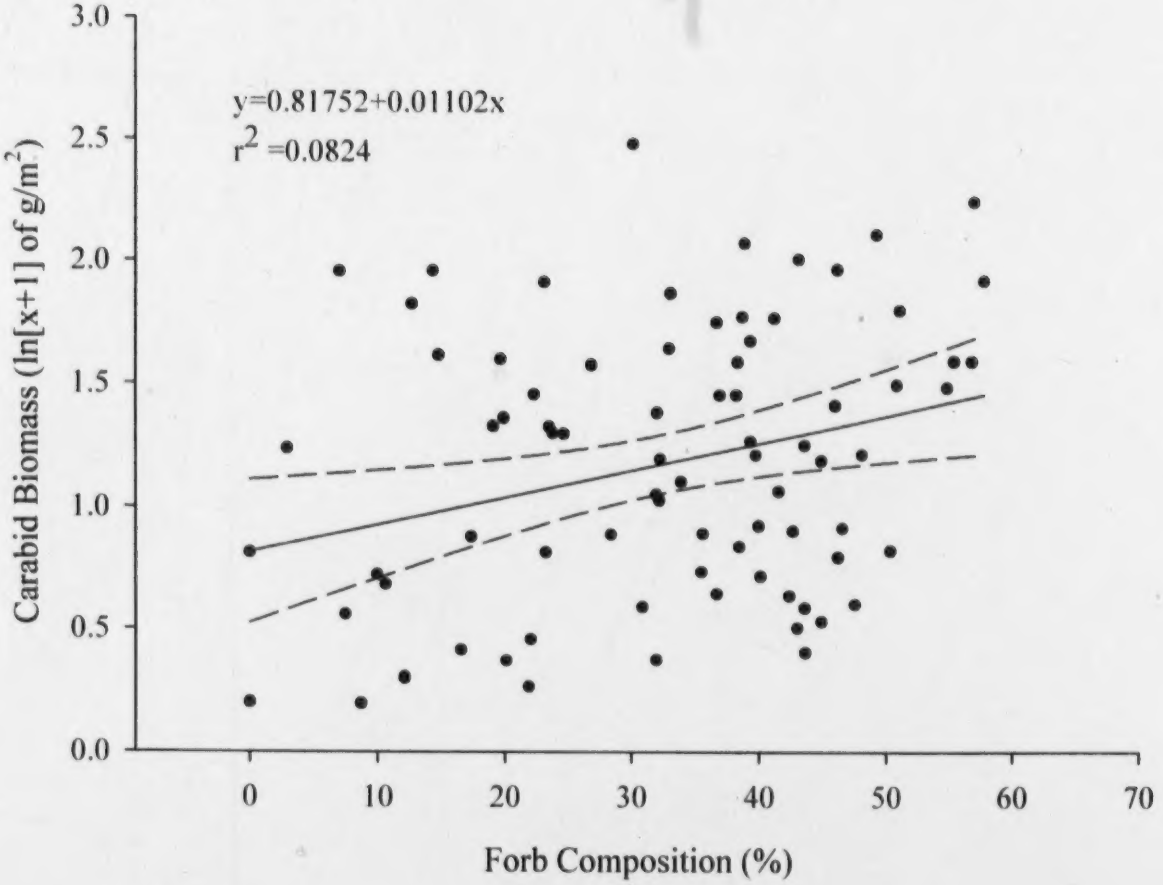
y=0.81752+0.01102x
r2 =0.0824
Carabid Biomass (ln[x+1] of g/m2)
Forb Composition (%)
3.0
2.5
2.0
1.5
1.0
0.5
0.0
0
10
20
30
40
50
60
70

Figure 2.19

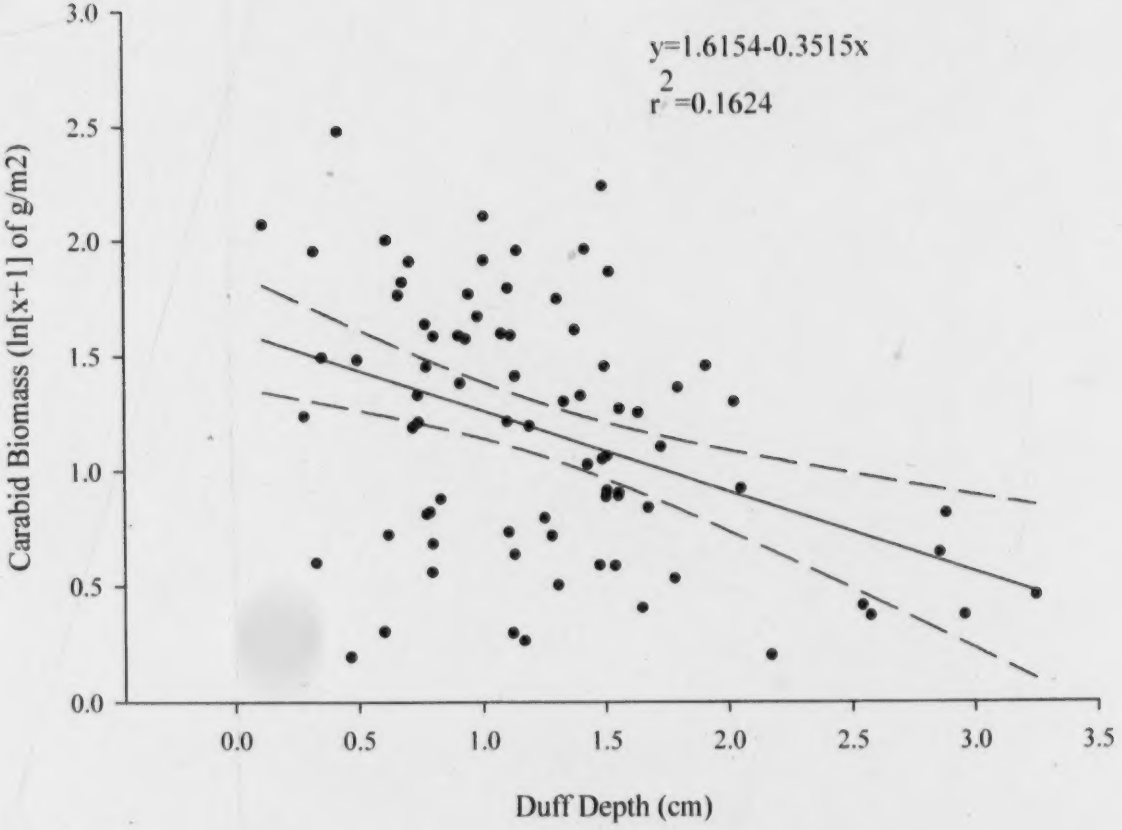
y=1.6154-0.3515x
r 2 =0.1624
Carabid Biomass (ln[x+1] of g/m2)
Duff Depth (cm)
3.0
2.5
2.0
1.5
1.0
0.5
0.0
0.0
0.5
1.0
1.5
2.0
2.5
3.0
3.5

Figure 2.20

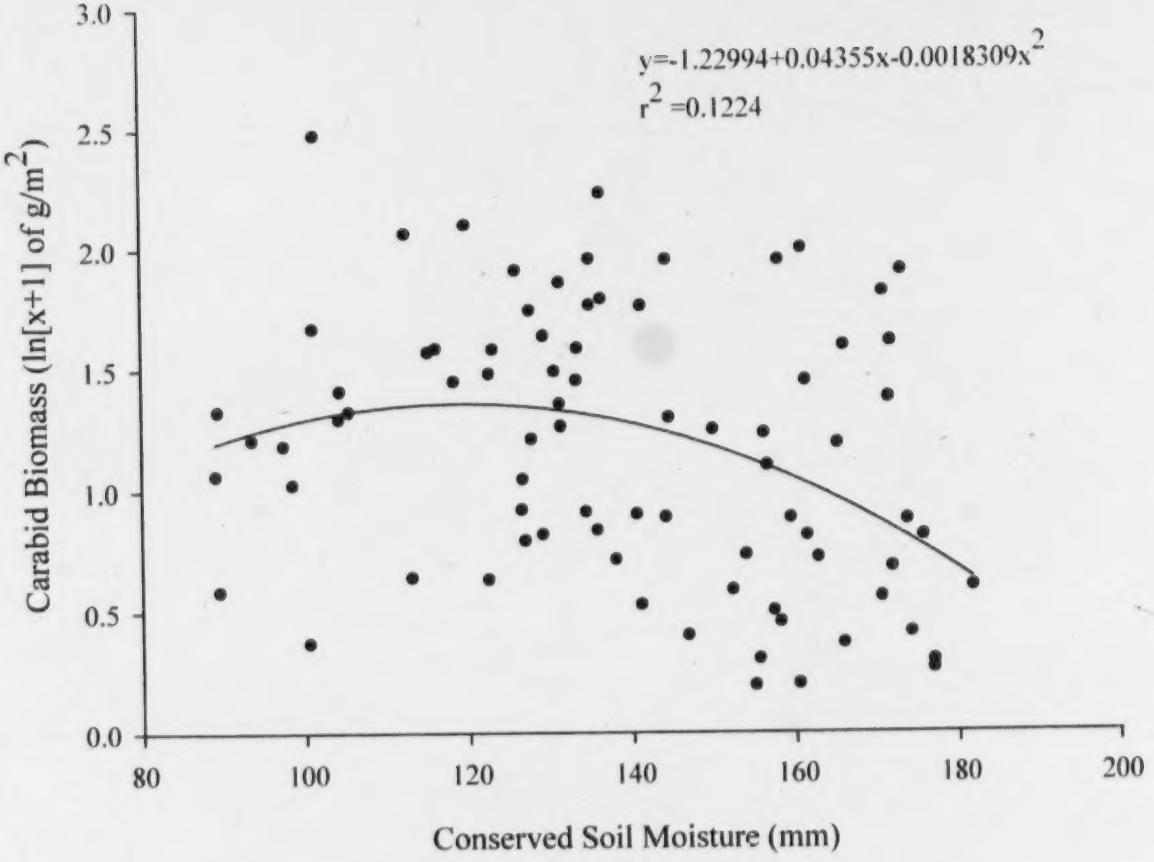
y=-1.22994+0.04355x-0.0018309x2
r2 =0.1224
Carabid Biomass (ln[x+1] of g/m2)
3.0
2.5
2.0
1.5
1.0
0.5
0.0
80
100
120
140
160
180
200
Conserved Soil Moisture (mm)

Figure 2.21

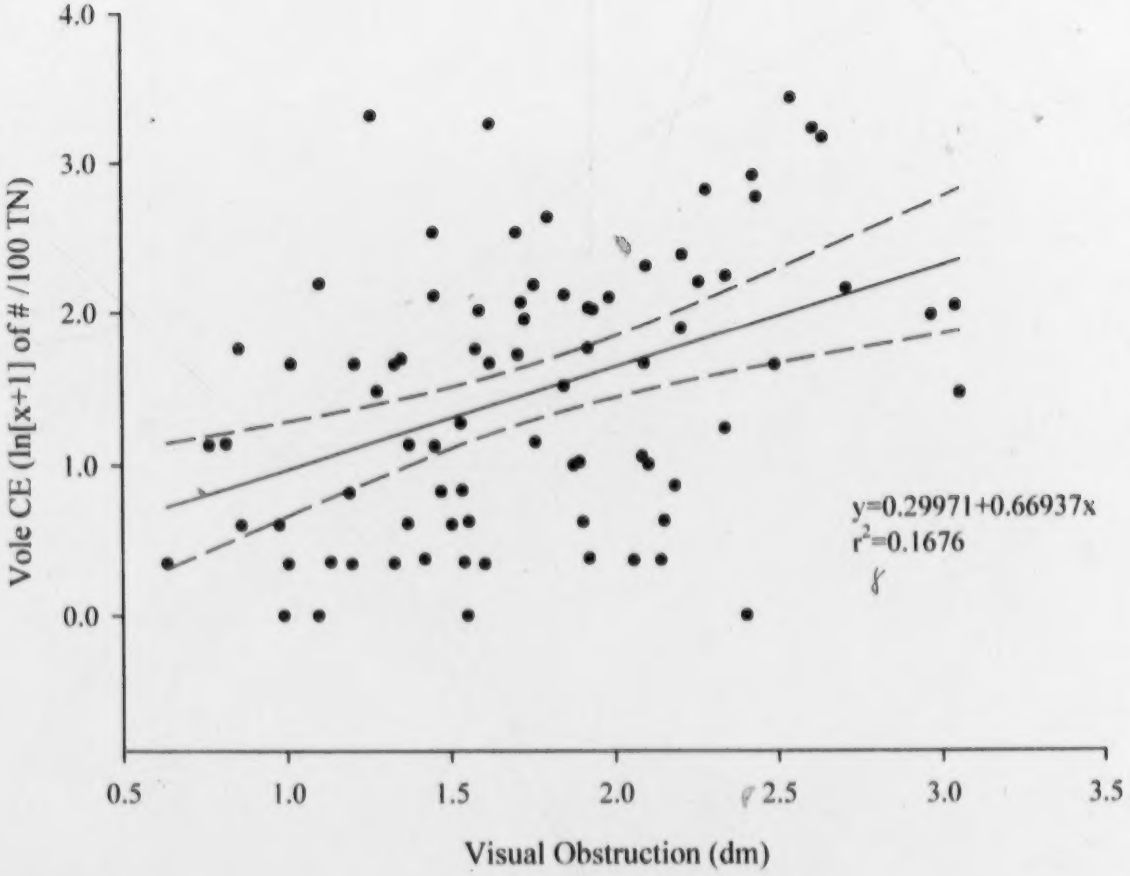
4.0
3.0
2.0
1.0
0.0
Vole CE (ln[x+1] of # /100 TN)
0.5
1.0
1.5
2.0
2.5
3.0
3.5
Visual Obstruction (dm)
y=0.29971+0.66937x
r²=0.1676

Figure 2.22

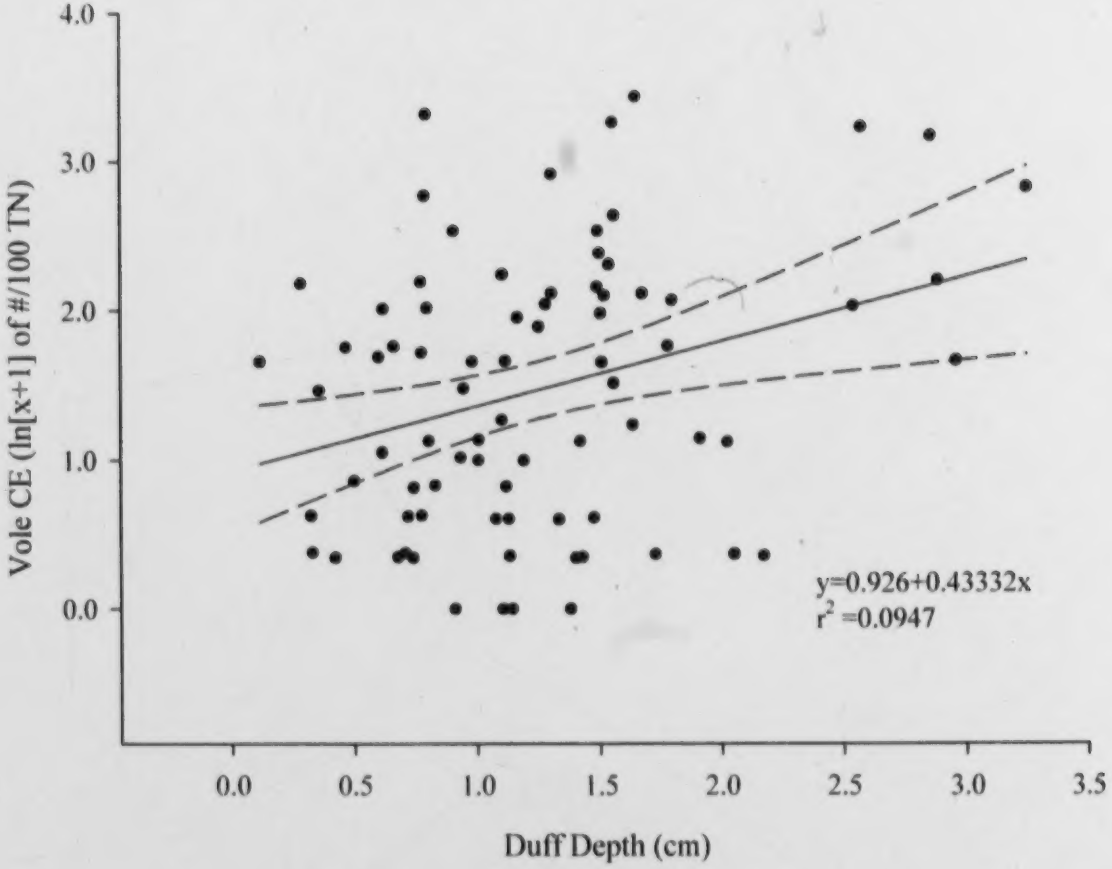
Vole CE (ln[x+1] of #/100 TN)
4.0
3.0
2.0
1.0
0.0
0.0
0.5
1.0
1.5
2.0
2.5
3.0
3.5
Duff Depth (cm)
y=0.926+0.43332x
r² =0.0947

Figure 2.23

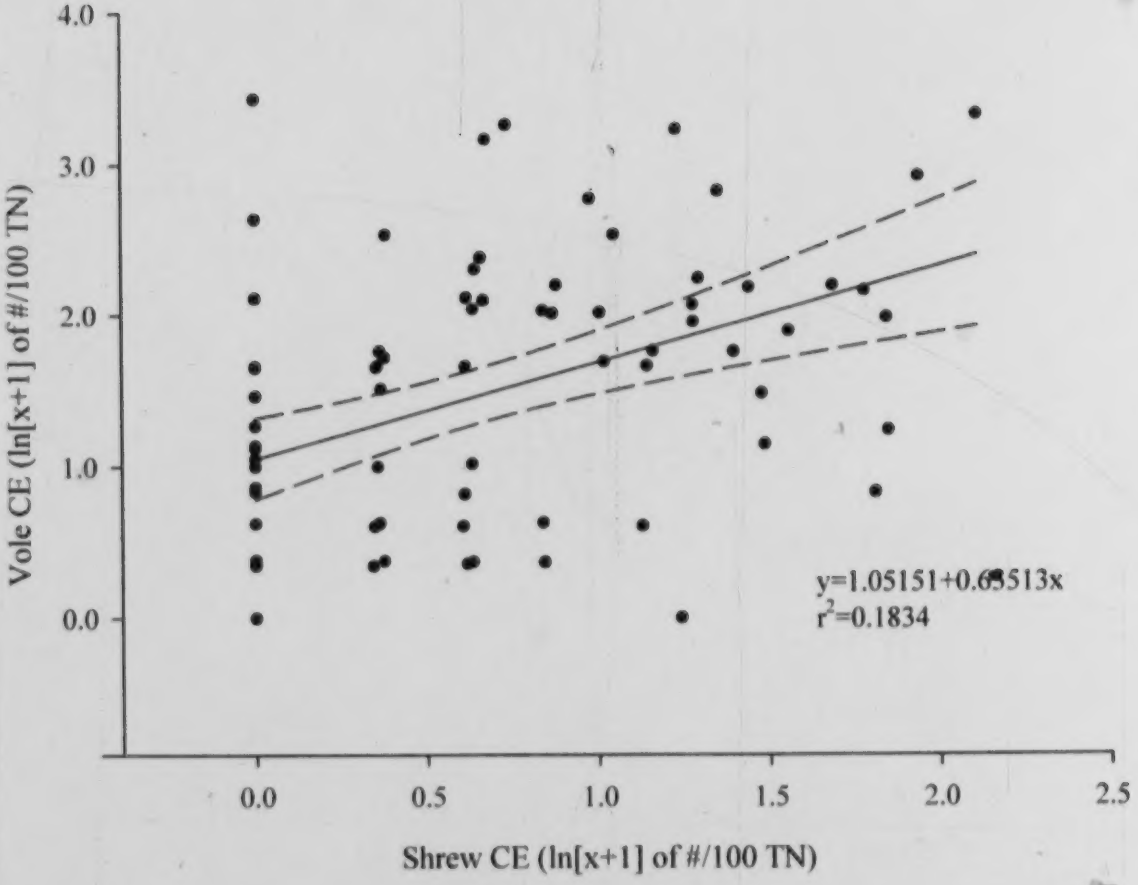
4.0
3.0
2.0
1.0
0.0
Vole CE (ln[x+1] of #/100 TN)
y=1.05151+0.65513x
r²=0.1834
0.0
0.5
1.0
1.5
2.0
2.5
Shrew CE (ln[x+1] of #/100 TN)

Figure 2.24

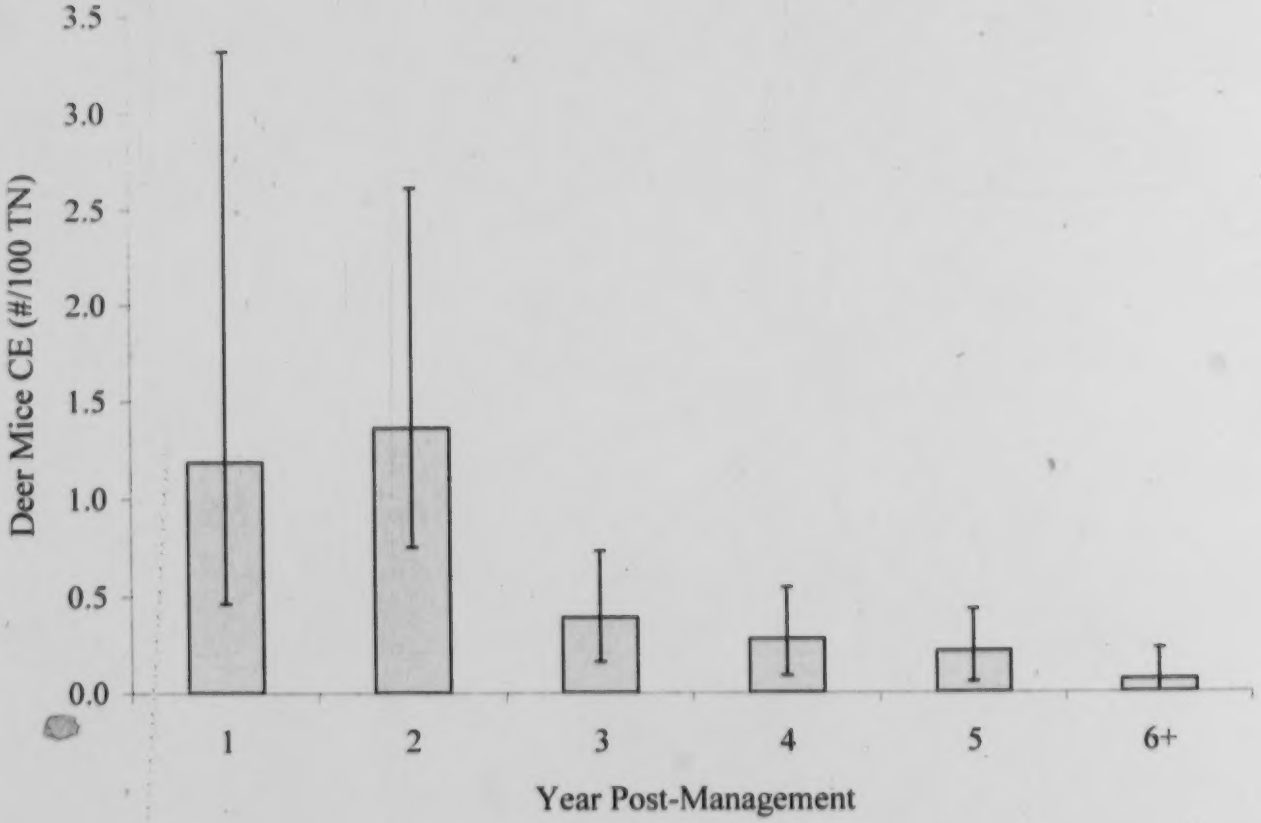
Deer Mice CE (#/100 TN)
3.5
3.0
2.5
2.0
1.5
1.0
0.5
0.0
1
2
3
4
5
6+
Year Post-Management

Figure 2.25

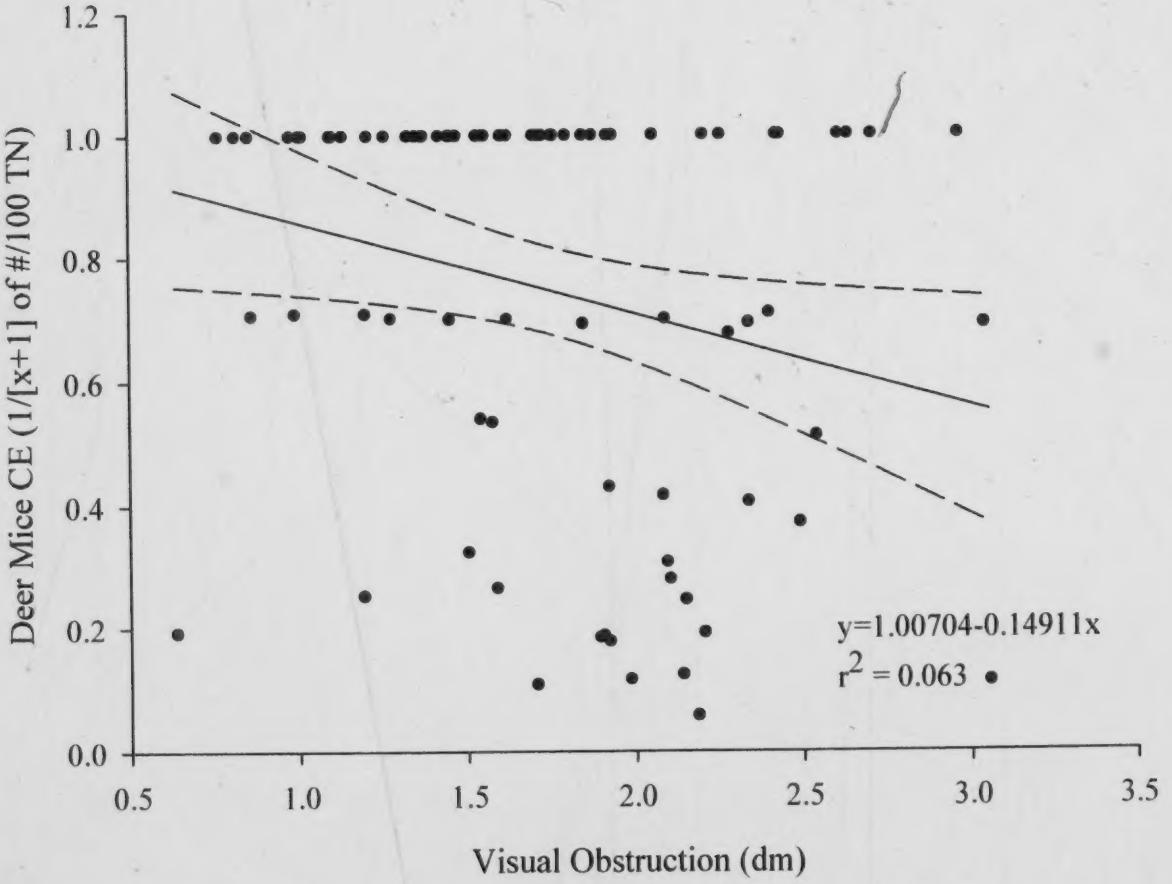
Deer Mice CE (1/[x+1] of #/100 TN)
1.2
1.0
0.8
0.6
0.4
0.2
0.0
0.5
1.0
1.5
2.0
2.5
3.0
3.5
Visual Obstruction (dm)
y=1.00704-0.14911x
r2 = 0.063

Figure 2.26

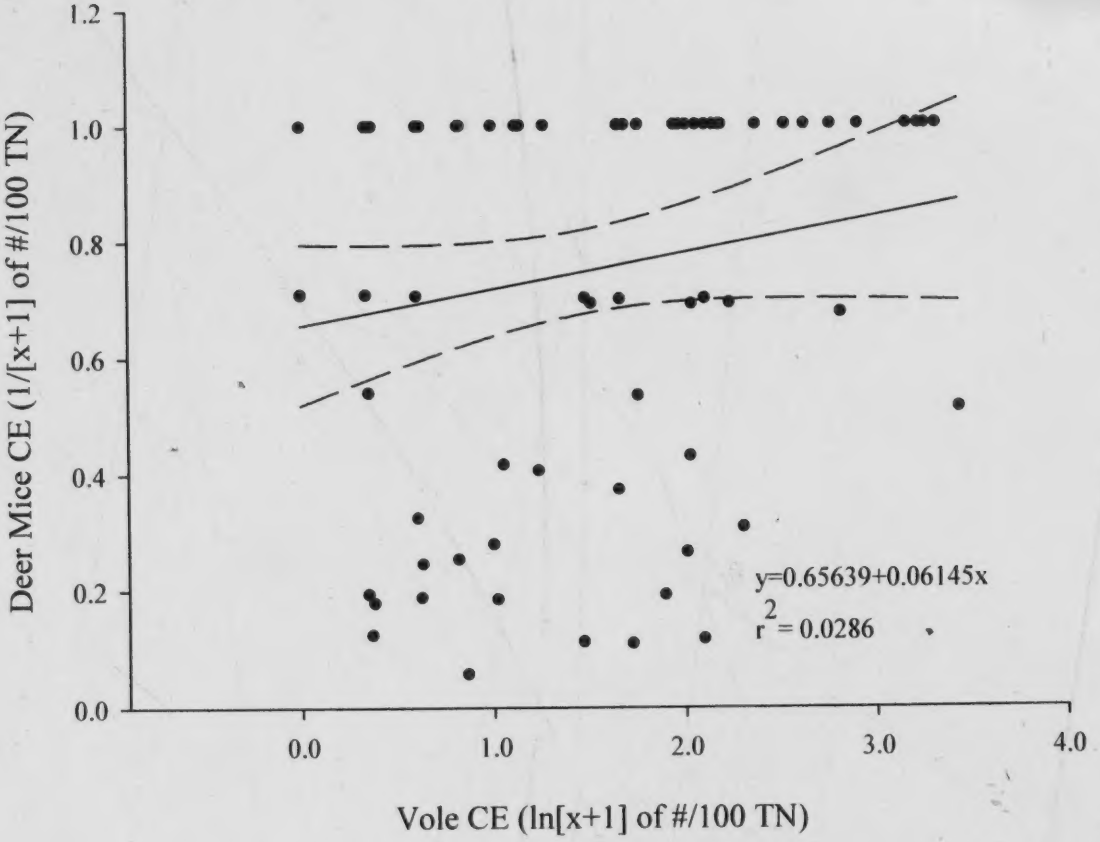
Deer Mice CE (1/[x+1] of #/100 TN)
1.2
1.0
0.8
0.6
0.4
0.2
0.0
0.0
1.0
2.0
3.0
4.0
Vole CE (ln[x+1] of #/100 TN)
y=0.65639+0.06145x
r 2 = 0.0286

Figure 2.27

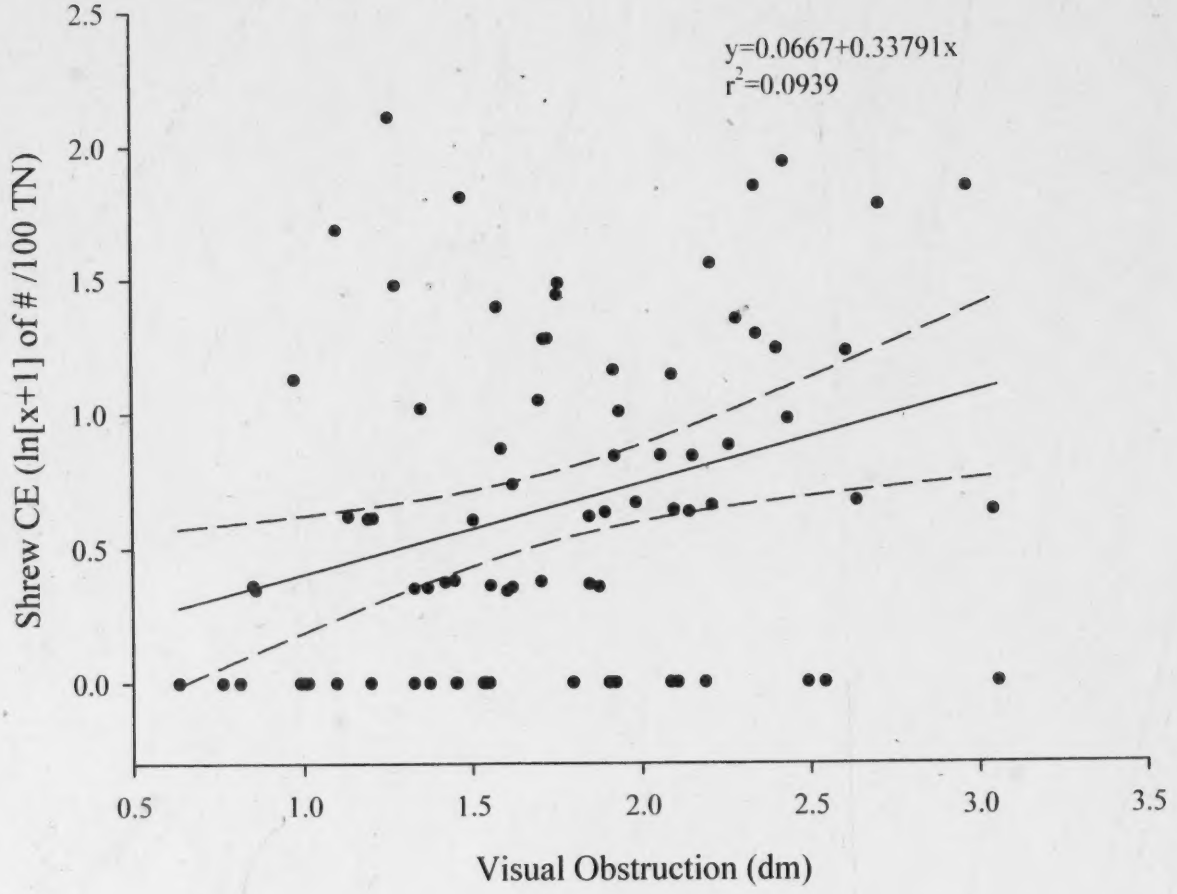
y=0.0667+0.33791x
r2=0.0939
Shrew CE (ln[x+1] of # /100 TN)
2.5
2.0
1.5
1.0
0.5
0.0
0.5
1.0
1.5
2.0
2.5
3.0
3.5
Visual Obstruction (dm)

Figure 2.28

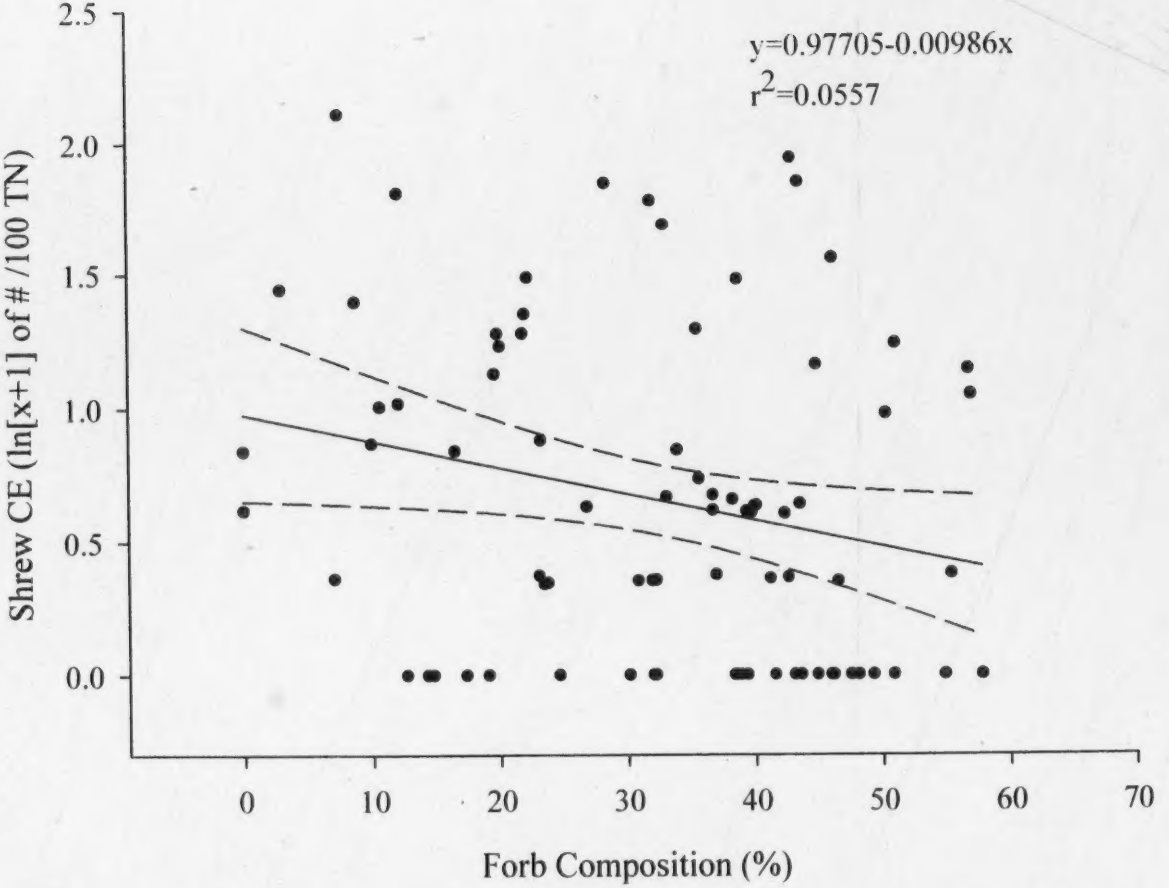
y=0.97705-0.00986x
r2=0.0557
Shrew CE (ln[x+1] of # /100 TN)
2.5
2.0
1.5
1.0
0.5
0.0
0
10
20
30
40
50
60
70
Forb Composition (%)

Figure 2.29

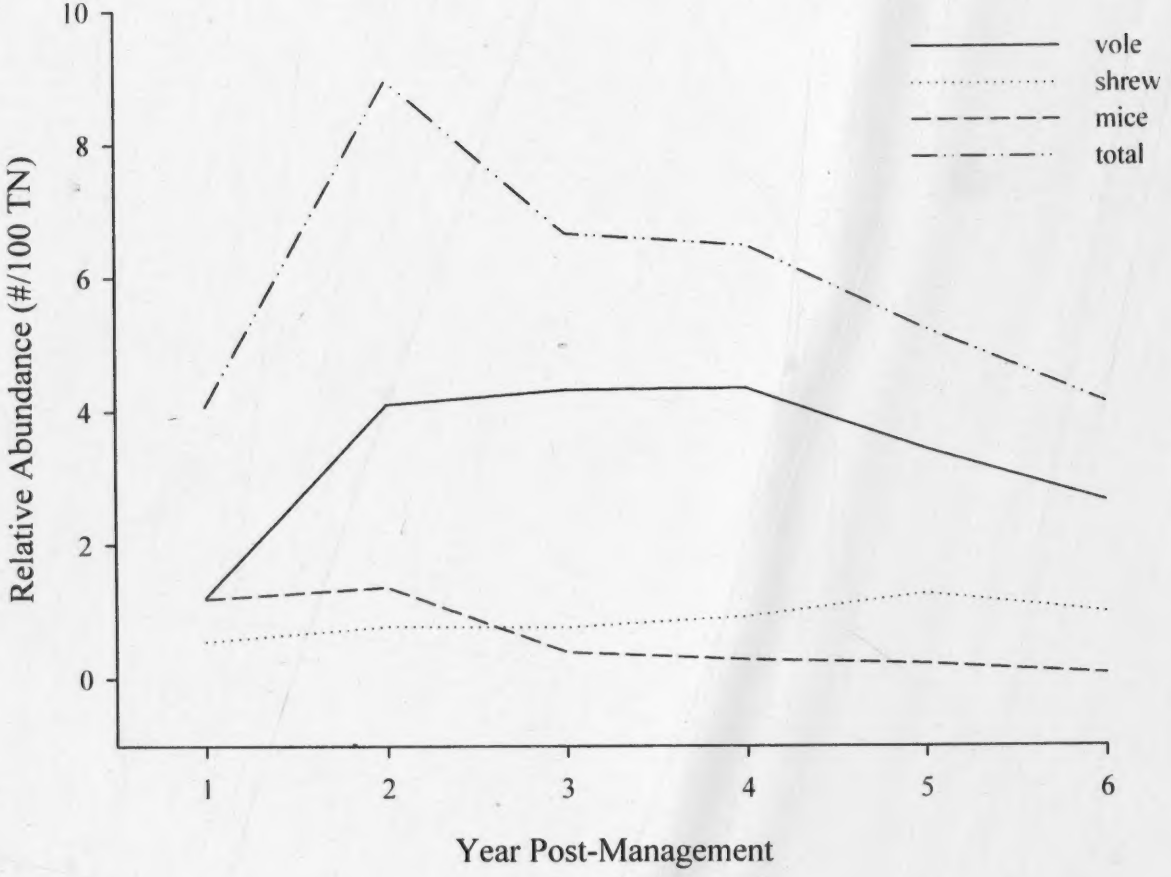
10
8
6
4
2
0
Relative Abundance (#/100 TN)
1
2
3
4
5
6
Year Post-Management
vole
shrew
mice
total

Figure 2.30

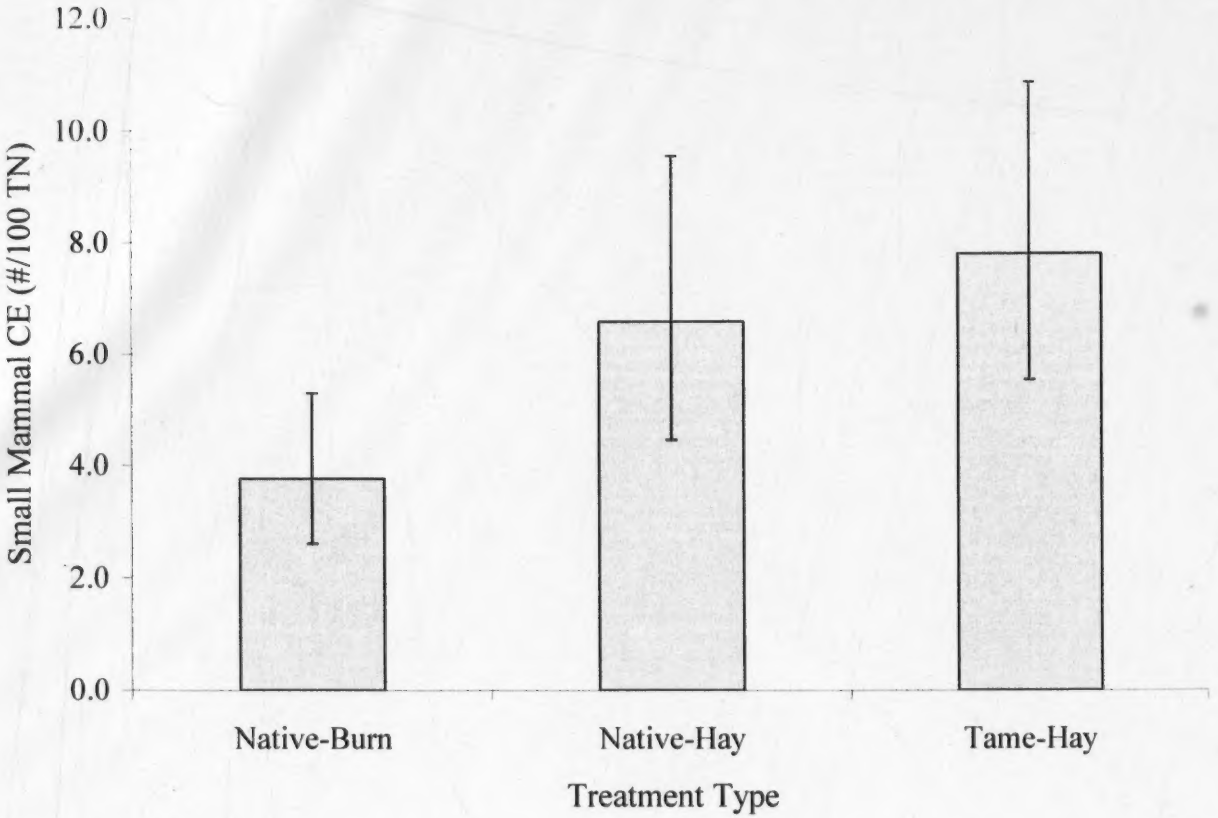
12.0
10.0
8.0
6.0
4.0
2.0
0.0
Small Mammal CE (#/100 TN)
Native-Burn
Native-Hay
Tame-Hay
Treatment Type

Figure 2.31

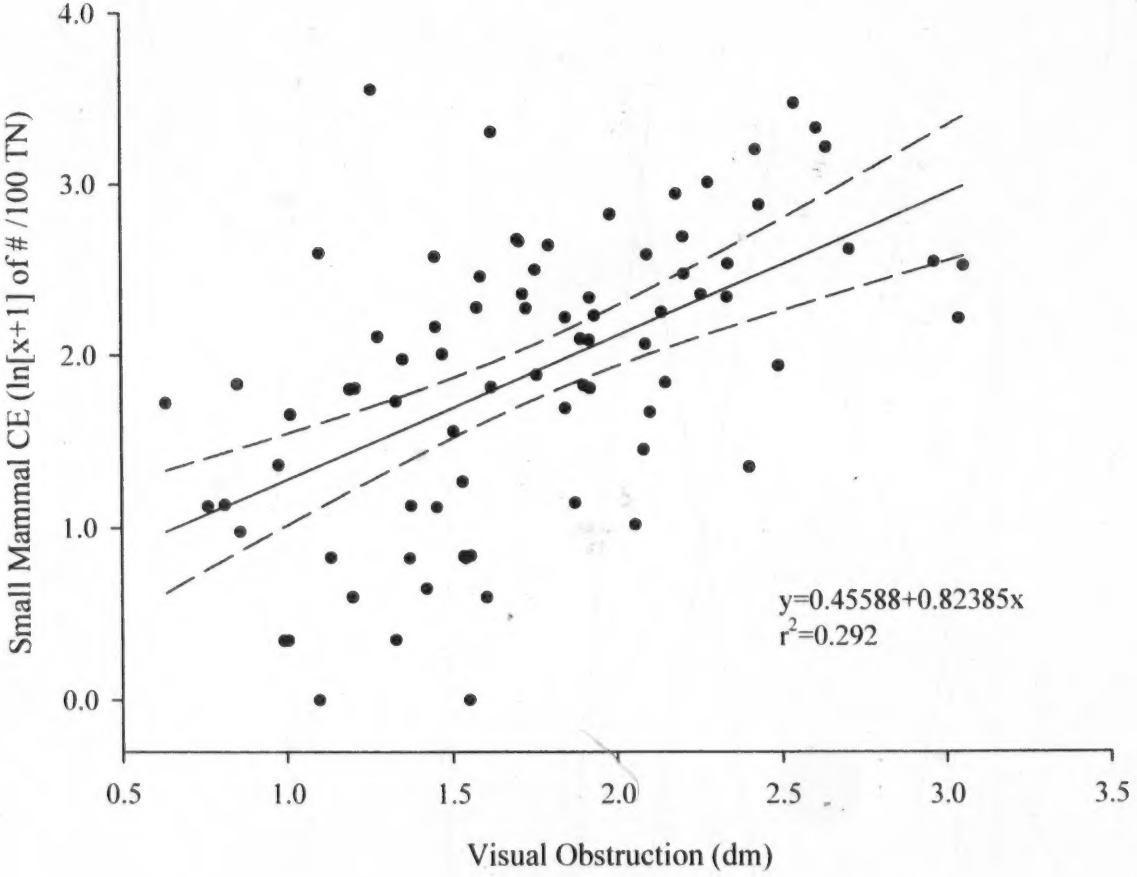
4.0
3.0
2.0
1.0
0.0
Small Mammal CE (ln[x+1] of # /100 TN)
y=0.45588+0.82385x
r²=0.292
0.5
1.0
1.5
2.0
2.5
3.0
3.5
Visual Obstruction (dm)

Figure 2.32

CHAPTER 3

Effects of Alternative Prey Abundance on Waterfowl Nesting Success within the Aspen Parkland of Saskatchewan and Manitoba[2]

Abstract. Nest depredation continues to be the main limitation for North American dabbling duck recruitment and, as such, recent focus has been directed at the potential for alternative prey species to buffer predation. Principle nest predators, such as red fox (*Vulpes vulpes*) and striped skunk (*Mephitis mephitis*), also readily select small mammals and terrestrial invertebrates. We located duck nests and sampled small mammals and beetles on 45 permanent cover fields managed as waterfowl nesting habitat in the Aspen Parkland of Saskatchewan and Manitoba during 2000 and 2001 to evaluate the role of alternative prey on duck nest survival. Using an extension of the Mayfield method, we modelled nest success probabilities with vole and beetle abundance, also including vegetation, landscape, and predator activity covariates in models. Akaike's Information Criterion, AIC, was used to select the most parsimonious model and average variable weights to determine individual variable importance. The best models included vegetation height, wetland area, and beetle and vole abundance parameters. We found a positive relationship between vole abundance and nest success; a negative relationship between beetle abundance and nest success.

Key Words: Akaike Information Criterion, AIC, permanent cover, waterfowl nesting success, alternative prey, Aspen Parkland, abundance, *Microtus pennsylvannicus*, meadow vole, beetle

[2] This chapter is formatted for submission to Ecological Applications.

INTRODUCTION

Historically, the majority of North American upland nesting duck production emanated from the grasslands and aspen parklands of the northern Great Plains of Canada and the United States (Baldassare and Bolen 1994). Much of this landscape, known as the Prairie Pothole Region (PPR), has been lost or severely fragmented due to agricultural conversion of natural grasslands to cropland (Sugden and Beyersbergen 1984; Miller and Nudds 1996), drastically reducing the amount of suitable upland waterfowl nesting habitat. While prairie nesting birds have evolved with native predators and are able to withstand high rates of predation without apparent effects on population size, predation may act as a secondary, exacerbating factor in the face of habitat alteration and subsequent changes in predator community structure (Cote and Sutherland 1997). Nest depredation continues to be one of the primary factors limiting waterfowl recruitment (Klett et al. 1988; Sargeant and Raveling 1992; Hoekman et al. 2002) but continued human alteration of the prairies has also altered predator abundance and distribution (Sargeant et al. 1993; Sovada et al. 1995), resulting in concentrations of predators, ducks, and alternate prey into the remaining grassland habitats. Conversion of cultivated lands to permanent cover is one method of ameliorating habitat loss. Through the North American Waterfowl Management Plan (NAWMP; U.S. Fish and Wildlife Service and Canadian Wildlife Service 1986) and Ducks Unlimited Canada (DUC), approximately 55,800 ha were converted to permanent cover between under this plan (Devries 2003).

The Alternative Prey Hypothesis (APH; Angelstam et al. 1984; Angelstam 1985) proposes that during high densities of primary prey species, the predation pressure on alternative prey species is relaxed and these alternative species benefit with increased

survival rates. Thereafter, as main prey densities decline, predators turn to the alternative prey whose survival rates decline with the increased pressure. In arctic and sub-arctic ecosystems, the abundance of alternative prey for mammalian and avian generalist predators has been found to be inversely related to predation rates on birds and nests (Angelstam et al. 1984; Korpimäki et al. 1990; Wegge and Storaas 1990; Leckie et al. 1998; Reif et al. 2001). Although the APH has been tested where there were fewer predator species and prey than the PPR, availability of food resources has been shown to influence predation rates on waterfowl nests (Byers 1974; Pehrsson 1986; Crabtree and Wolfe 1988). Of the 20 nest predators identified by Sargeant et al. (1993), red fox (*Vulpes vulpes*), striped skunk (*Mephitis mephitis*), American crow (*Corvus brachyrhynchos*), and badger (*Taxidea taxus*) were particularly important within the PPR with abundance varying spatially and temporally (Johnson et al. 1989). These principle predators are generalists with diets consisting of small mammals, birds, birds' eggs, insects, and amphibians. Red foxes are particularly important due to their ability to take both hen and eggs, causing an imbalanced sex ratio and loss of renesting potential (Johnson and Sargeant 1977). However, skunk predation on ground nesting birds is thought to be mainly incidental (Vickery et al. 1992; Crabtree and Wolfe 1988) through focal foraging at wetland edges (Phillips et al. 2003) with foraging patterns altered by invertebrate availability and abundance (Greenwood et al. 1999).

The objective of this study was to investigate the role of alternative prey, specifically vole and beetle abundance, on duck nest success in managed permanent cover habitats. We examined nest success and prey abundance in agricultural uplands previously converted back to native or tame grass/legume stands. Established stands had

been managed through either burning or haying 1 to 8 years prior to study commencement. Furthermore, because a variety of vegetation and landscape parameters have been indicated in nest success, vegetation characteristics (Kaminski and Weller 1992), wetland area (Greenwood et al. 1995), conserved soil moisture (Boyd 1981; Bethke and Nudds 1995), extent of cropland (Greenwood et al. 1995), patch size (Clark and Nudds 1991), and predator activity were also examined.

Study areas

Sites were primarily located within the Aspen Parkland Ecoregion of Saskatchewan and Manitoba although seven sites were located within the Moist Mixed Grassland Ecoregion of Saskatchewan (Figure 3.1) as classified by the Canadian Land Classification System (Ecological Stratification Working Group 1995). This prairie region is comprised of nearly level glacial till, glacial lake, or glacial river plains with numerous hummocky and ridged moraines throughout. Elevation ranges between 500 and 600 m with the exception of hummocky morainal deposits such as the Touchwood Hills, Saskatchewan that can exceed 730 m. A humid continental climate prevails with mean July temperatures of 18.0 °C and mean January temperatures of -18.9 °C; mean annual precipitation in Saskatchewan is 420 mm (Acton et al. 1998). Dark brown and black chernozemic soils dominate the landscape with dark gray chernozems occurring in boreal transition areas and solonetzic soils occurring in association with inland drainage basins (Acton et al. 1998). Although several major buried aquifers are present within the region, blanket aquifers are important sources of groundwater. Numerous isolated wetlands dot the landscape, ranging from ephemeral ponds, remaining wet only a few weeks in spring, to semi-permanent deep open-water ponds. Many permanent ponds and

lakes are present in the region with saline lakes and ponds occurring where inland drainage systems are dominant.

Through Ducks Unlimited Canada (DUC) and the North American Waterfowl Management Plan (NAWMP; U.S. Fish and Wildlife Service and Canadian Wildlife Service 1986), parcels of previously cultivated lands were converted to permanent waterfowl dense nesting cover (DNC) between 1985 and 1995. Conversion consisted of seeding with either a native grass/legume mixture or an introduced (tame) grass/legume mixture.Seed mixes varied by region to accommodate soil types and moisture regimes with the more common native species being: northern, western, or slender wheatgrass (*Agropyron spp.*), green needle-grass (*Stipa viridula*), side-oats grama grass (*Bouteloua curtipendula*), sand dropseed (*Sporobolus cryptandrus*), prairie clover (*Dalea spp.*), black-eyed susan (*Rudbeckia hirta*), meadow blazingstar (*Liatrus ligulstylis*), bergamot (*Monarda fistulosa*), cut-leafed anemone (*Anemone multifida*), prairie coneflower (*Ratibida columnifera*), hedysarum (*Hedysarum spp.*) and western snowberry (*Symphoricarpos occidentalis*). Introduced mixes may include tall, intermediate, or slender wheatgrass (*Agropyron spp.*), wild rye grass (*Elymus spp.*), brome grass (*Bromus spp.*), creeping red fescue (*Festuca rubra*), and alfalfa (*Medicago spp.*). Stands tended to exhibit a high degree of structural diversity and heterogeneous density within and between fields.

Projects were managed post-seeding to control weed and shrub invasions and to maintain/improve stand health and vigour. For the purposes of the study, projects managed 1 to 8 years prior to this study through either burning or haying were included. No management occurred subsequent to this burn/hay event. Projects were categorized

by treatment (TRT) as native-burn, native-hay, or tame-hay with an age (i.e. growing seasons) since management identifier (year post-management [YPM]). For the most part, DUC does not manage tame stands through burning therefore there was no tame-burn category. YPM categories were not constant for the duration of the study since stand age increased by one year between sampling years (Figure 3.2). YPM 6 to 8 were pooled into one 6+ YPM category because there were few or no replicates within treatments. Forty-one projects were sampled in 2000; 42 were sampled in 2001. Project sizes (area of dense nesting cover) ranged from 6 to 62 ha (refer to Appendix A for project characteristics).

METHODS

Geographic and Climatic Characteristics

Geographical location (latitude and longitude) and conserved soil moisture (CSM) were included as environmental variables. Project latitude and longitude, in the form of decimal degrees were taken from the DUC Project Database and multiplied to achieve an index of project location (GEO). CSM was used as an index of wetness regimes and wetland conditions (Boyd 1981; Bethke and Nudds 1995) where precipitation records for the 21 months preceding May 1, 2000 and 2001 were used to calculate soil moisture according to the Williams and Robertson (1965) formula:

$$CSM = 0.36A+[0.37B–0.2(0.36A)]+0.13\ C+\{0.30D–0.2[0.36A+(0.37B–0.2(0.36A))+0.13C]\}$$

where A = total precipitation during August through October in year $t - 2$,

B = total precipitation during November in year $t - 2$ through April in year $t - 1$,

C = total precipitation during May through October in year $t - 1$,

D = total precipitation during November in year $t - 1$ through April in year t.

Prior to calculating CSM, a proximity analysis using ArcGIS (Environmental Systems Research Institute, Redlands, California) software was conducted to determine weather stations within 30 km of each project. We used monthly precipitation records from 50 stations and calculated CSM per weather station for 2000 and 2001 (Monthly Precipitation Records, Atmospheric Environment Service, Downsview, Ontario). Individual project CSM was generated as an average of the CSM for the three or four nearest weather stations, taking into consideration the weighted inverse distance from the project to each station.

A natural precipitation gradient occurs across the region such that moisture, and subsequently CSM, increases in a south-eastwardly direction. We tested for correlation between CSM and project location with a Pearson Correlation Matrix. A marked difference in precipitation was noticed between 2000 and 2001 therefore paired t-tests were used to test for between-year difference in CSM.

Adjacent Land Use Characteristics

Land use was characterized for the 4 quarter sections adjacent to each project. Areas were visited in July, categorized to 1 of 13 land uses (see Appendix B for land use categories), and delineated on aerial photographs. Grain crops were identified as either fall- or spring-seeded whereas other cereal or oil crops were identified to crop type (i.e. flax, corn, pea, or canola). Non-cropped areas were categorized as native areas left idle, native areas with grazing, planted cover, or hay fields. Fallow areas (bare soil) and areas with buildings or recreational infrastructure (e.g. farms, churches, graveyards, or golf

courses) were categorized separately from surrounding land uses. Air photos were digitized by using AutoCad (AutoDesk, Inc., San Rafael, California) software to determine aerial extent of each use category.

Land use characteristics were consolidated into two functional land use groups: IDLE (non-cropped areas) and CROP (cropped or fallow areas). Percent area per group was calculated and further arcsine transformed (Table B.24, Zar 1999) to accommodate underlying normality assumptions (Zar 1999). CROP was chosen to include in modelling because landscape-level crop information is easier to quantify and readily available through Agriculture Statistics Division of Statistics Canada.

Planted Nesting Cover Area

The amount of planted nesting cover (also referred to as dense nesting cover [DNC]) was variable between projects. To determine the area of coverage per project, the extent of nesting cover was delineated on air photos and area calculated through digitizing using AutoCad.

Wetland Area

Wetlands within each project and a circumscribed 400 m strip were classified according to Stewart and Kantrud (1971) in July of each year. Wetlands were delineated on aerial photographs during classification based on extent of wetland vegetation. There was no perceived change in the extent of wetland vegetation between sampling years, therefore, wetland area was determined through digitizing the 2000 data. Only wetlands with permanency class III, IV, and V (seasonal, semi-permanent, and permanent) were included in the analysis.

Wetland size, density, and distribution are variable across the prairie region. To provide an index of wetland area (WA) per project, total wetland area was divided by the area of DNC within the project. WA was natural log transformed (ln[x+1]) to more closely represent a normal distribution. Correlation between WA and CSM was tested with a Pearson Correlation Matrix.

Vegetation Characteristics

Vegetation at all projects was sampled three times (May 22 – May 28; June 12 – June 18; July 3 – July 9) per year at permanently established stations. The number of sampling stations depended on project size: 9 stations for DNC less than 32 ha, and 16 stations for DNC exceeding 32 ha. Locations of sampling stations (Universal Transverse Mercator [UTM] projection) were randomly generated using SPANS GIS (PCI Geomatics, Richmond Hill, Ontario). A grid of systematic random points was established with a Global Positioning System (GPS) throughout each project. To reduce station-placement bias, once at the approximate point, a random number of paces between 1 and 20 and direction (0 - 360°) were generated on a standard mathematical calculator. A spike was driven into the soil at the sampling point and a willow stake set 4 m north to aide in station relocation. Both spike and willow stake were flagged with orange surveyors tape.

Sampling sessions consisted of four sets of measurements at each station: visual obstruction readings (VOR; Robel et al. 1970), duff depth (DUF), maximum height (MXHT), and vegetation composition. VOR was estimated by placing a 14 dm pole demarcated at 0.5 dm intervals at the sampling point and recording the last mark not visible through the vegetation from 1 m above ground level at a distance of 4 m from the

pole. Four VOR were recorded, one in each cardinal direction. Duff depth, described as the dead and/or decaying horizontal vegetative matter on the soil surface, was measured 30 cm from the sampling point in each cardinal direction. A plastic ruler was inserted vertically through the duff layer until the ruler contacted the soil layer; duff depth was recorded to the nearest 0.5 cm. Maximum vegetation height was determined as the tallest piece of new or residual vegetation within a 15 cm radius from the Robel pole. A slotted clear plastic disc (30 cm diameter) was slid down the pole until the first piece of vegetation was encountered; height was measured to the nearest 0.5 dm. Vegetation composition was measured as comparative percent grass, forb, and shrub aerial cover within one square metre.

Recently managed projects initially had substantially less cover than projects idled for several years. Seasonal growth rates were not similar and, as such, did not show parallel density trajectories throughout the summer sampling periods. We opted to use an overall summary of conditions by averaging the three sampling period means to generate an index of field-level conditions. Because percent grass and percent forb composition were ipsitive, only forb cover was included in the analysis. Percent forb composition (FORB) was arcsine transformed to accommodate underlying normality assumptions (Table B.24, Zar 1999). Percent shrub was negligible and was not included in the analysis.

Small Mammal Abundance

A single trapping session for small mammals was conducted between July 16 and August 12, 2000 and July 8 and August 4, 2001. Small mammals were sampled at 25 sampling stations in a 5 X 5 grid layout (Figure 2.4) (Innes and Bendell 1988; Jones et al.

1996). Sampling stations, consisting of two Museum Special snap-traps (Woodstream Corp., Lititz, Pennsylvania), were spaced 10 m apart. Grid placement was randomly generated for each field with the selection constraint that adjacent habitats (i.e. wetlands, tree stands, crops, and roads) were more than 50 m from any side of the grid. The northwest corner of each grid was marked via GPS units to facilitate grid re-establishment in 2001. Traps were baited with a peanut butter/rolled oat mixture and serviced for five consecutive mornings (between 0600 and 1200 hrs). Due to the distance between projects, between 9 and 12 projects were sampled during a trapping period with these traps relocated to new sites at the close of the five night trapping period.

Specimens were stored in 120 ml Whirl-Pak® bags; large individuals (e.g. *Spermophilus spp.* and *Thomomys spp.*) were stored in Zip-Lock® bags. Date, location, species, and sex (if determinable) were recorded at the time of collection. Confirmation of species identification, sex, breeding condition, weight, and standard body measurements were determined at the Museum of Zoology, University of Alberta where the specimens are stored. Sampling procedures were approved by the Manitoba Wildlife Animal Care Committee (MWACC #2000-01) and the Faculty of Agriculture, Forestry and Home Economics Animal Policy and Welfare Committee (FAPWC #2001-10D).

A measure of relative abundance (number of small mammals caught relative to the trapping effort) was generated by calculating corrected catch effort (CE) as a measure of trap success per 100 TN (Nelson and Clark 1973; Holroyd and Van Tighem 1983; Beauvais and Buskirk 1999). Relative abundance estimates assume capture numbers represent a relatively constant but unknown relationship to the total population size although the number of captures further depends on animal activity, movement, and

trapper skill (Krebs 1994). If these conditions are reflected in trapping success, catch effort becomes an index of abundance measuring relative density of animals. CE was calculated following the Nelson and Clark (1973) formula that accounts for decreases in trapping efficiency due to traps being sprung by animals other than the species of interest, weather, vegetation, or any other cause.

$$CE = (100 \cdot A)/(TU - S/2)$$

where A = number of individuals of the desired species caught;

TU = P•I•N (number of trapping units);

P = number of trapping intervals;

I = length of trapping interval;

N = number of traps;

S = number of traps sprung by all causes.

Non-mammalian specimens were included in the analysis only as sprung traps to determine corrected CE. Due to low captures of some species, individuals were grouped together as functional units such that vole species (*Microtus pennslyvanicus* and *Clethrionomys gapperi*) were grouped and shrew species (*Sorex haydeni*, *Sorex cinereus*, *Sorex hoyi*, *Sorex arcticus*, and *Blarina brevicauda*) were grouped. Pocket gophers and thirteen-lined ground squirrels were considered incidental captures as they were non-target species. Jumping mice occurred in too few numbers and were not included in the analysis. Total small mammal CE was generated by pooling vole, deer mice, and shrew captures. Deer mice abundance data were reciprocal ($1/[x + 1]$) transformed while vole, shrew, and total abundance data were natural log transformed ($\ln[x + 1]$) to accommodate normality assumptions.

Small Mammal Biomass

Biomass, as a quantitative measure of community-level production, was calculated per species group (deer mice, voles, and shrews) and total small mammal biomass as mean wet weight (kg) of individuals per hectare (Brower et al. 1998).

$B = \sum W/A$

where $\sum W$ = sum weight of individuals (grams)/1000;

A = total area sampled (area of trapping grid = 1600 m^2 or 0.16 ha).

Biomass calculations were transformed with a reciprocal (1/[x+1]) transformation for deer mice and natural log (ln[x+1]) transformation for voles, shrews, and total small mammals. A Pearson Correlation Matrix was used to determine the degree of correlation between abundance and biomass estimates.

Beetle Abundance

Invertebrates were sampled from May 3 – July 14, 2000 and May 7 – July 12, 2001 with pitfall traps (modified design based on Morrill 1975 and Schmidt and Lockwood 1992). Traps were located 5 m north of the odd-numbered permanent vegetation sampling stations in each project and serviced every 7 to 10 days. Traps consisted of an outer plastic beverage cup (473 ml; perimeter 27.5 cm) with a removable inner cup trimmed down to a height of 4.5 cm. They were placed with the lip flush with ground level and the inner liner ¾ filled with a saturated salt solution (Figure 2.3). During servicing, traps were emptied, rinsed with water and re-filled with solution, or replaced if damaged. Samples within each field were combined at the time of collection and identified to Order. Specimens were stored in 70% isopropyl solution.

Beetle specimens were sorted, identified to Family, counted, and air dried. Weights were determined per Family. Voucher specimens identified by Dr. George E. Ball, Entomologist, are housed at the Strickland Museum of Entomology, University of Alberta.

Although 11 arthropod Orders were represented, only beetles were included in the analysis as they constituted the single largest percentage of captures (over 40%). Pitfall traps reflect activity and density of surface-dwelling insects, but captures are also a function of microclimate, trap material, preservative used, and beetle assemblage (Southwood 1978; Spence and Niemelä 1994; van den Berghe 1992; Waage 1985). Beetle catch per unit effort (CPUE) was calculated as total captures per 100 trap nights (TN) to determine an index of relative abundance.

$$CPUE = (A/TN) \cdot 100$$

where A = number of individuals.

Silphids (burying and carrion beetles) and Carabids (ground beetles) comprised the bulk of captures. However, Silphids are attracted to odours emitted from carrion, therefore, traps already containing captured individuals may have acted as a lure. As such, Silphid counts and weights were excluded from abundance and biomass calculations. Relative abundance was generated separately for Carabids.

Beetle Biomass

Numbers of individuals reflect density but do not provide qualitative accounts of production capacity since individuals and Families vary greatly in size. Indices of beetle biomass (excluding Silphidae) and Carabidae biomass were calculated as total dry weight of individuals per square meter (Brower et al. 1998).

$B = \sum W/A$

where $\sum W$ = sum weight of individuals (grams);

A = total trap area sampled (m^2; total number of TN•area of each trap [56.745 cm^2] /10,000). Carabidae and total beetle abundance and biomass estimates were natural log transformed (ln[x+1]) to accommodate normality assumptions. A Pearson Correlation Matrix was used to determine the degree of correlation between abundance and biomass estimates.

Predator Activity

Avian and mammalian predators were surveyed in 2001 to determine their respective activity levels. For avian predators, two road counts were conducted 21 days apart between May 29 and July 4 along a 16 km transect on available roads circumscribing each project (Fuller and Mosher 1981). Multiple projects shared one transect if projects were less than 10 km apart or if individual transects would have intersected or overlapped given the available roads. Surveys were conducted between 0630 and 0730 hours by two observers travelling at 25-40 km/h and were completed in approximately 30 minutes. Species, location along the route, and activity (i.e. perching, nesting, in flight) of those individuals occurring within 800 m of the road were recorded. Identification was aided by binoculars when necessary.

Avian use of areas and subsequent detection during road counts are influenced by weather, seasonal changes in bird activity, terrain, vegetation, infrastructure (e.g. telephone poles, fences, and buildings), road type, and human activity (Fuller and Mosher 1981). Avian predator activity (BA) was calculated as the number of individuals observed per kilometre:

$$BA = A/D$$

where A = number of individuals observed;

D = total number of kilometres surveyed.

Because Franklins's gulls have not been implicated as nest predators, only corvids, hawks, ring-billed gulls and California gulls were included in the analysis (Sargeant et al. 1993). We were interested in field-level estimates, therefore, abundances were averaged over the two surveys to generate an index of relative abundance. To accommodate normality assumptions, BA was natural log transformed (ln[x+1]).

Mammalian predators were surveyed between July 9 and August 4 in conjunction with small mammal sampling. Two scent-stations per project were monitored for five consecutive nights. The stations were located 800 m apart or at opposite ends of a project for smaller projects. Stations, consisting of a circular pad 1 m in diameter, were placed in areas of sparse vegetation in the ditch, right-of-way, or along a fireguard. The pad was cleared of rocks, levelled, and re-surfaced with sifted soil. A Fatty Acid Scent attractant disk (Pocatello Supply Depot, Pocatello, Idaho) was attached to a centrally located wooden stake (Linhart and Knowlton 1975). Surface soil was resifted each day for optimal track imprint. Tracks were identified to species according to Stokes and Stokes (1999). Species presence was recorded per night.

Visits to scent-stations vary among species and is influenced by type of lure used, terrain, time of operation, placement within territory, and animal age and sex (Seber 1982; Wilson and Delahay 2001). Mammalian predator activity level was generated as the number of visits per sampling effort, calculated as visitation rate (VR) according to Linhart and Knowlton (1975):

$VR = (A/N) \cdot 1000$

where A = number of visits;

N = number of operable station nights.

Number of operable station nights excludes those in which a station was damaged by adverse animal activity or weather. Unknown tracks were included in the analysis except for domestic dog and domestic cat tracks.

Nest Searches

Beginning on May 8, DNC was searched three times per year at 21-day intervals to locate and assess fates of duck nests. Projects were searched in the same sequence per interval and year using a cable-chain drag similar to Higgins et al. (1977) and pulled with two all-terrain vehicles. Search direction and established tracks were maintained to minimize disturbance. Searches were conducted between 0730 and 1300 hours to maximize the likelihood that hens would be attending nests (Gloutney et al. 1999). Following Klett et al. (1986), a nest was defined as ≥ 1 egg tended by a hen when found. Incubation stage was determined by candling at least two eggs per visit (Weller 1956). Assuming 1 egg was laid per day and partial nest depredation had not occurred prior to when a nest was located, we estimated nest initiation date for nests found during laying as date found less the number of eggs. Initiation date for incubated nests was determined as date found less the sum of incubation stage and clutch size. Estimated hatch date was calculated by adding clutch size and average incubation period (by species) to date found.

Nests were marked by placing a willow switch (1 – 1.5 m length) tied with a piece of orange surveyors tape 4 m north and locations were recorded on aerial photographs. Nests were revisited every 7 – 10 days until fate was determined. Nest fate, and if nest

failed to hatch, suspected cause of failure, were recorded at final visit. A nest was deemed successful if ≥ 1 egg hatched, as determined by presence of shell membranes (Klett et al. 1986), or ducklings were present in nest bowl. Nests were deemed unsuccessful if all eggs were destroyed, missing, abandoned, or non-viable. Nests which incurred damage or were partially depredated when found, abandoned subsequent to the first visit, or abandoned due to investigator activity were excluded from analysis.

Analysis

Estimates of predator activity given management, vegetation characterstics, and landscape covariates were made using Akaike's Information Criterion AIC_c, adjusted for small sample size because the ratio of sample size to parameters was less than 40 (Akaike 1973, Burnham and Anderson 1998):

$$AIC_c = (-2 \log \text{likelihood}) + 2K + (2K[K+1] / [n-K-1])$$

where log likelihood is the maximized log-likelihood value, K is the total number of estimated parameters (including all independent variables, dependent variables, intercept, random factor, and residual variance), and n is sample size. The model with the lowest AIC_c is considered the most parsimonious or "best" model providing a quantitative assessment of the "strength of evidence" given the data (Burnham and Anderson 1998). Inferences were made on candidate models in which the ΔAIC_c (AIC_c - minimum AIC_c) values were ≤ 2 (Burnham and Anderson 1998).

Five *a priori* management models were examined for their ability to predict predator activity among treatment types and YPM. Fifteen *a priori* vegetation models were then examined for their ability to predict activity at projects in response to VOR, duff depth, maximum height, and forb composition (refer to Appendix E for model

construction). Because we were examining the predator community, 18 landscape covariates were examined for their importance to overall predator activity. Although CSM and project location were highly correlated, we opted to include project location as a landscape variable because predator communities tend to differ geographically (Sargeant et al. 1993). Other landscape covariates included linear and quadratic terms for wetland area, DNC area, cropped area, and prey abundance or biomass estimates. Models within 4.0 ΔAIC_c units from the management and vegetation suites were combined with landscape covariate and covariate combinations to develop an exploratory suite of models. A null model, that used only the intercept, random, and residual parameters to model the response variable, was included in each model suite as a measure of model fitness. Akaike model weights were used to determine the weight of evidence for each model (Burnham and Anderson 1998):

$$w_i = \exp(-0.5 * \Delta_i) / \sum^{R}_{r=1} \exp(-0.5 * \Delta_i)$$

where $\exp(-0.5 * \Delta_i)$ is the relative likelihood of a model and $\sum^{R}_{r=1} \exp(-0.5 * \Delta_i)$ is the sum of all relative likelihoods within the candidate set of models. An assessment of model strength relative to other models was achieved using model weight ratios (w/w_i) where the best model has a weight ratio of 1. To quantify the importance of each variable over the set of competing models, average variable weights (w_v) for each variable were calculated by summing the model weights in which a specific variable occurs and dividing it by the number of models in which it occurs (Burnham and Anderson 1998). Statistical analysis was performed with SAS, version 8.2 (Littell et al. 1996; SAS Institute 1999).

Using an extension of the Mayfield method (Manly and Schmutz 2001), nest survival probabilities were modelled as a function of prey abundance, management effects, and continuous covariates within non-linear mixed models (SAS' Proc NLMIXED), allowing both fixed and random effects to have non-linear relationships with survival (Wolfinger 1999). Each nest was considered an experimental unit where nest fate over the period of observation (Y_{ij}) was treated as binomially distributed with index parameter 1 and survival probability (π_{ij}) between 0 and 1. We defined the following:

Y_{ij}	fate of the nest over the specified interval; 1=successful, 0=failed
t_{ij}	length of the i^{th} interval in the j^{th} project
$X_{1ij}, X_{2ij}, \ldots, X_{Kij}$	categorical and continuous covariates
ε_j	random effect associated with the j^{th} project
DSR_{ij}	daily survival rate associated with the i^{th} interval in the j^{th} project, where $\text{logit}(DSR_{ij}) = \beta_0 + \beta_1 X_{1ij} + \ldots + \beta_1 X_{Kij} + \varepsilon_j$
π_{ij}	probability of surviving the interval, $\pi_{ij} = DSR_{ij}^{t_{ij}}$

Successful nests would have a fate $Y_{ij} = 1$ with a single survival interval t_{ij} = hatch date – date found. Because we could not know the exact date of failure for unsuccessful nests, two records were generated for each nest evaluating a survival interval and failure interval. The survival interval represented the period of time during which the nest was under observation and known to still be viable ($Y_{ij} = 1$ and t_{ij} = date of second last visit – date found); failure interval represented the interval during which the nest was known to fail ($Y_{ij} = 0$ and t_{ij} = date of the last visit – date of the second last visit). Because we felt field level conditions not specifically modelled may have affected survival, we specified

project as a random effect. To estimate DSR and survival probability, we used a maximum likelihood approach where DSR was fit to a logistic regression model with a specified linear predictive function which accounted for covariates of interest and random effects. Probability of a nest surviving the interval (π_{ij}) was defined as a function of DSR and interval length.

The most parsimonious model structures were chosen based on Akaike's Information Criterion (AIC; Burnham and Anderson 1998):

AIC = (-2 log likelihood) + 2K

To address the primary question of the effect of prey abundance on nest success, two sets of data were analysed. The first set contained 2000 and 2001 data whereas the second set contained only 2001 data but included predator activity measures. We initially examined nest survival probabilities in relation to year, management effects, vegetation parameters, landscape covariates, and prey abundance. *A priori* and exploratory models were developed and models selected similar to what was previously described with the exception that 33 landscape models were fit to the data. Within the landscape model suite, linear and quadratic functions of wetland area, DNC area, CSM, adjacent cropland, and beetle abundance were fit with and without vole abundance. Appendix F provides a list of model structures and covariates used.

RESULTS

Geographic and Climatic Characteristics

CSM exhibited a strong inverse correlation with latitude (r = -0.848) and longitude (r = -0.872) with study site locations. An increase in CSM occurs with

decreased latitude and increased longitude. In other words, locations within the south-eastern portion of the study site (e.g., the Minnedosa area of Manitoba) had higher moisture levels than the north-western portion (e.g., the Saskatoon area).

Wetland Area

WA was not correlated to CSM ($r = 0.023$), therefore, both variables were included in the analyses.

Vegetation Characteristics

Vegetation characteristics that differed between year, among treatments, and by YPM are evaluated in Chapter 2. VOR, maximum height, and forb composition decreased between sampling years. Density was greatest in Native-Hay and Tame-Hay treatments but the increase with CSM varied among treatments. Density and maximum height were lowest in the first YPM, highest in YPM 2 and 3, and intermediate in subsequent YPM. Native-Burn treatments had the least amount of forb composition (13%) with other treatments having considerably greater forb content (33 – 51%). Duff depth was also shallowest in Native-Burn treatments.

Small Mammals

We generated a total of 10,000 trap nights in 2000 for 1,038 small mammals captures; 10,250 trap nights were generated in 2001 with 462 captures. The most commonly captured species were meadow vole (*Microtus pennslyvanicus*; 61%), deer mice (*Peromyscus maniculatus*; 14%), and masked shrews (*Sorex cinereus*; 13%; Table 3.1). The effects of management techniques, vegetation characteristics, and landscape variables on vole and total prey abundance were evaluated in Chapter 2. Abundance and biomass estimates were highly correlated and exhibited a substantial decrease

(approximately 55%) between sampling years. Abundance was lowest in Native-Burn projects and highest in YPM 2 – 4; vegetation density was the primary explanatory variable of abundance such that abundance increased with vegetation density.

Beetles

Of the 13 Coleopteran Families represented by captures, Silphidae and Carabidae constituted the majority of individuals with 52% and 37% respectively (Table 3.2). Once Silphidae were excluded, Carabidae constituted 78% of captures and 85% of biomass (range 40 – 100%). Twenty-eight Carabid species are represented among the 56 specimens identified (refer to Appendix D for a list of species).

The effects of management techniques, vegetation characteristics, and landscape variables on beetle and carabid abundance and biomass were evaluated elsewhere (Maile et al. In Prep.). Total beetle abundance and biomass were highly correlated whereas carabid abundance and biomass were not. An increase of approximately 45% occurred between sampling years for total beetle CPUE. Beetle abundance decreased with duff depth, increased with forb composition, and increased at intermediate CSM levels. Carabid abundance and biomass were greatest in the second and third YPM. A quadratic relationship with CSM and vegetation density further explained variability. Carabid biomass doubled between sampling years; biomass decreased with greater duff depth and CSM.

Avian Predator Activity

The most common avian species observed were American crow (*Corvus brachyrhynchos*), black-billed magpie (*Pica pica*), red-tailed hawk (*Buteo jamaicensis*), and Franklin's gull (*Larus pipixcan*) (Table 3.3).

Management models could not adequately predict activity in the proximity of projects because the null model was a better model than management models (Table 3.4). The linear and quadratic maximum height models were the most parsimonious vegetation models with the quadratic model having a considerably larger model weight ratio (w/w_i = 1) compared to the linear model (w/w_i = 0.6624). Activity increased with maximum height (r^2 = 0.1501; β_{MXHT} = -0.743, SE = 0.468; β_{MXHT}^2 = 0.0796, SE = 0.0433; Figure 3.5). A third model, DUF+MXHT was also within 2.0 ΔAIC_c units but was nearly equivalent to the null model, indicating this combination was a poor model. CROP was chosen as the better landscape model where avian abundance was inversely related to amount of cropland proximal to the projects (r^2 = 0.1736; β_{CROP} = -0.00858, SE = 0.00296; Figure 3.6). Combinations of the better predictor covariates from each model suite resulted in the MXHT+MXHT^2+CROP+GEO model as the most parsimonious with no close competitors based on model weight ratios. A negative relationship existed between geographic location and avian activity (r^2 = 0.1885; β_{GEO} = -0.00043, SE = 0.00029; Figure 3.7). Projects in western Saskatchewan had the lowest avian activity and Manitoba the highest. Overall, activity was highest in areas of less cropland and on the eastern side of the study area.

Mammalian Predator Activity

Tracks of mammalian predator species were observed over 350 operable trap nights. Red fox (*Vulpes vulpes*), coyote (*Canis latrans*), and weasel (*Mustela spp.*) were most commonly recorded at scent-stations (Table 3.5).

Mammalian activity differed among management treatments where Native-Hay had the lowest visitation rate (Figure 3.8; Table 3.6). Among the vegetation models, the

MXHT+FORB model was the most parsimonious with no close competitors. FORB had a somewhat higher variable weight ratio (w/w_i = 1.0) than MXHT (w/w_i = 0.7587). Activity was inversely related to percent forb composition (r^2 = 0.2096; β_{FORB} = -10.351, SE = 2.582; Figure 3.9) but positively related to maximum height (r^2 = 0.0914; β_{MXHT} = 102.33, SE = 34.538; Figure 3.10). GEO was the best landscape covariate model. Mammalian predator activity was inversely related to geographic location (r^2 = 0.2766; β_{GEO} = -0.461, SE = 0.13; Figure 3.11). The MXHT+FORB+GEO combination was selected as the most parsimonious exploratory model with no close competitors. Activity appeared to be greatest in Manitoba and where vegetation was tallest with less activity recorded next to projects with greater forb composition.

Nest Success

Total DNC area searched summed to 1,060 ha in 2000 and 1,125 ha in 2001. Of 959 duck nests monitored, 871 were used in the analysis (Table 3.7). Nests of blue-winged teal (*Anas discors*; 52%), mallard (*Anas platyrhynchos*; 18%), northern shoveler (*Anas clypeata*; 17%), and gadwall (*Anas strepera*; 10%) constituted the majority found. American widgeon (*Anas americana*), green-winged teal (*Anas crecca*), northern pintail (*Anas acuta*), and lesser scaup (*Aythya affinis*) nests were encountered infrequently but included in the analysis.

Nest success for the study was estimated at 13% with the greatest success occurring in Native-Burn projects in YPM 1 and 6+ (Figure 3.12). However, management effects could not be considered good predictors as they were no better than the null model (Table 3.8). The most parsimonious vegetation model was the quadratic MXHT model. Nest success appeared to be lowest at intermediate maximum height

values (β_{MXHT} = -2.077, SE = 0.721; β_{MXHT}^2 = 0.181, SE = 0.0619; Figure 3.13). Five landscape models were within 2.0 ΔAIC units with linear or quadratic functions of beetle abundance, vole abundance, and wetland area recurring in competing models. Examination of 175 exploratory models revealed six competing models where beetle abundance, maximum height, and wetland area were present in each candidate model. The quadratic maximum height term carried the greatest average variable weight, w_v = 0.02381, followed by the quadratic wetland area term, w_v = 0.01996, and beetle abundance, w_v = 0.0173. While the vole abundance parameter also occurs in candidate models, the average variable weight, w_v = 0.0148, is half that of the maximum height term and considerably less than the beetle abundance term. Nest success appeared to be greatest where the wetland to DNC ratio was approximately 1.5:1 hectares (β_{WA} = -1.569, SE = 1.778; β_{WA}^2 = 0.334, SE = 0.138; Figure 3.14). While nest success appeared to have a positive relationship with vole abundance (β_{VOLE} = 0.107, SE = 0.1047; Figure 3.15), nest success decreased sharply as beetle abundance increased (β_{BEET} = -1.043, SE = 1.213; Figure 3.16). The combination of maximum height, wetland area, beetle and vole abundance appear to be the best explanation for the levels of nest success we encountered.

Using the 2001 data set, management and vegetation models were not good predictors by themselves although 5 vegetation models were within 2.0 ΔAIC units (Table 3.9). Two models were selected from the landscape suite with the quadratic wetland area model being the most parsimonious. Examination of 203 exploratory models failed to reveal a single best model, instead 15 models were within 2.0 ΔAIC units. Wetland area and maximum height appeared to be the most important covariates

(w_v = 0.0086 and w_v = 0.0085 respectively) with avian predator activity (w_v = 0.0063), beetle abundance (w_v = 0.0061), and DNC area (w_v = 0.006) having nearly equivalent weight values but somewhat less important. For the 2001 data, nest success appeared to be higher where avian predator activity was greater (β_{AVE} = 0.45, SE = 0.323; Figure 3.17) and within medium-sized projects (β_{DNC} = -0.0101, SE = 0.00767; Figure 3.18). With the exception of avian predator activity, the important covariates explaining nest success are similar to results from the full data set.

DISCUSSION

Modelling nest success is helpful for exploring complex situations and allows us to examine nest survival probabilities under a variety of conditions. Our nest success models lead us to believe several parameters, in association with each other, were the predominant influences on nest success. It appeared vegetation height, wetland area, and prey abundance were principle factors in explaining nest success. Although management effects seemed to be poor explanatory variables for nest success, differences among YPM and treatments were apparent. Nest success appeared to be greatest in Native-Burn projects and within the first and 6+ YPM. Projects with intermediate levels of maximum height and either low or high wetland area relative to DNC exhibited low nest success. Furthermore, projects in which beetles were abundant showed the lowest nest success whereas projects with high vole abundance exhibited the greatest success. While there was a positive relationship between success and vole abundance, the negative relationship with beetle abundance may have overridden the positive effects of vole abundance since

YPM 2 through 4 exhibited the greatest vole and beetle abundance yet had the lowest success.

Prey availability not only varies temporally within and between years but also spatially. In response, generalist predators may switch foraging strategies to take advantage of prey availability. Northern studies have shown the importance of small mammals in nesting success where canid or avian predators switch from preying on ground nests to small mammals during peak rodent years. In Scottish moorlands, red fox switched from rodents to red grouse (*Lagopus lagopus scoticus*) as small mammal abundance decreased (Leckie et al. 1998) and positive effects of alternative prey on nest success has been shown for goose nests (Bêty et al. 2002), Oldsquaw (*Clangula hyemalis*; Pehrsson 1986), and black grouse (*Tetrao tetrix*; Angelstam et al. 1984). This switch, common among generalist canid predators, also appears to hold for some generalist avian predators (Korpimäki et al. 1990; Rief et al. 2001). While most alternative prey studies have concentrated on northern systems, recent studies in California (Ackerman 2002) and Saskatchewan (Pasitschniak-Arts et al., In Prep.) also found positive indirect effects between vole abundance and duck nest success.

The positive relationship between vole abundance and nest success from the current study supports recent Californian and Canadian studies. However, neither of these studies addressed terrestrial invertebrate abundance although skunks were considered a prominant nest predator. Skunks tend to feed on a variety of foods (e.g. carrion, insects, and nestling rodents) in the early part of the nesting season (May – June) but become primarily insectivorous in July when terrestrial insect abundance and mobility peak (Crabtree and Wolfe 1988). Skunk depredation of ground nesting birds is

thought to be incidental (Vickery et al. 1992) with focal foraging occurring at the wetland-upland interface, regardless of landscape composition (Phillips et al. 2003). Focal foraging at wetlands has also been suggested for raccoons (*Procyon lotor*; Greenwood 1981; Greenwood 1982). In contrast, red fox appear to preferentially select core areas and wetland edges only in areas with little (10 – 15%) grassland cover (Phillips et al. 2003). Our findings suggest greater wetland area relative to permanent cover corresponded to higher nest success but where the wetland to upland ratio exceeds 1.5:1 hectares, nest success decreased. Abundance of ponds has been found to have a positive influence on duck production (Krapu et al. 1983; Greenwood et al. 1995) although Reynolds et al. (2001) and Beauchamp et al. (1996) found no evidence that nest success was associated with number of wet basins/area or conserved soil moisture. Foraging in areas with little wetland area may result in prey at wetland edges being quickly exhausted, forcing predators to spend more time navigating between wetlands or seeking prey in the uplands. Wilson and Bromley (2001) found while arctic fox (*Alopex lagopus*) sightings did not differ between years, the amount of time foraging in goose nesting areas decreased during high lemming densities and Leckie et al. (1998) suggested red foxes switched habitats when prey abundance decreased. Similarly, skunk use of upland areas is year-dependent and skunks may be able to recognize profitable patches (Lariviere and Messier 2001). Concentration of foraging along the wetland interface would minimize skunk foraging in the upland areas as long as prey availability remains high. Because many of the projects sampled had less than optimal wetland to upland area, predators most likely had regular forays into the uplands. High beetle abundance in the uplands

would be beneficial to predators but could result in increased chance of encounters with duck nests.

When the optimal wetland to DNC ratio was exceeded, nest success decreased, possibly as a result of small patch size and resulting proximity of nests to wetland edge. Most dabbling ducks, with the exception of Northern Pintail, preferentially nest in close proximity to wetlands (Kaminski and Weller 1992). In a recent parkland spatial analysis, mallards tended to preferentially select small patches and nest closer to wetlands even though success was lowest under these conditions (Howerter 2003). In 2001, amount of DNC appeared to be as important as beetle abundance. Predators may be able to locate and effectively search small nesting habitats (Fritzell 1978; Krasowski and Nudds 1986; Clark and Nudds 1991), partially explaining the low nest success we encountered within small projects. While no clear relationship between patch size and nest success has been defined, Sovada et al. (2000) found small patches to have lower mean DSR. And although Burger et al. (1994) suggest predation rates increased in large prairie remnants, Kantrud (1993) found success to be highest in large CRP (Conservation Reserve Program) patches. Furthermore, Howerter (2003) suggested large patches are important for early (May) nesters in the Canadian parklands. This is contrary to our findings within the 2001 data set that success was also low in large patches of DNC. Variability tends to be the rule rather than the exception for nest success and a partial explanation for the inconsistency may be predator composition or age structure of the predator community in association with patch size, particularly if predator numbers are greater on larger tracts. Without large predators such as the coyote to control meso-predator (medium sized animals such as skunks and badgers) numbers, multiple small- and medium-sized

predators can co-exist within larger patches, often causing compensatory or increased predation pressure (Goodrich and Buskirk 1995; Estes 1996; Henke and Bryant 1999; Sovada et al. 2001). At large spatial scales, particularly in areas of small amounts of cropland or where predator removal still exists, the number of large predators may be insufficient to control meso-predators. Additionally, proximity of natal dens and burrows to nesting habitat will also influence predation pressure since females tend to forage closer to the den site during lactation (Lariviere and Messier 1998). Natal coyote and skunk dens as well as active badger burrows were observed on several projects but since these were not systematically recorded we can not determine how important this factor was to nest success. Furthermore, many known resident predators did not appear during surveys and, as such, our predator indices were likely underestimated. Future studies should include several censusing techniques to accurately depict predator communities.

Patch size and vegetation complex may be important in determining predator search capabilities and nest site selection has been linked to vegetation density and height (Kaminski and Weller 1992). Within our study, vegetation height in association with wetland area and alternate prey was important in determining nest success. Taller, denser vegetation has often been advocated as providing better lateral and vertical nest concealment (Naugle et al. 2000) with greater spatial heterogeneity forcing predators to expend more time searching for nests (Bowman and Harris 1980). Protection is afforded from aerial depredation as well as impeding mammalian predator movement, especially in dense cover where red fox movements tend to be more tortuous compared to cropland (Phillips et al. 2003). However, correlations between vegetation measures and success do not always exist (Elser and Grand 1995; Ackerman 2002) plus Clark and Nudds (1991)

contend that the importance of concealment is often dependent on predator community. Our predator acitivity estimates agree with the spatial variability of mammalian and avian predators across the PPR encountered by Greenwood et al. (1995).

The positive relationship between avian predator activity and nest success was unexpected because American crows are among the most important nest predators (Johnson et al. 1989). Higher predator activity levels were recorded in Manitoba where the abundance of shelterbelts and enhanced dispersion of perching sites may have produced larger avian communities (Murphy 1993). CSM was highly correlated with geographic location and while moisture did not appear to be an important parameter, we did notice nest success was elevated in areas of greatest moisture (i.e. in Manitoba). Therefore, we feel avian activity and greater nest success were likely independent rather than causal and because the mammalian activity relationship was also positive, our activity indexes may not accurately reflect the true predator community and variable importance. Because known resident skunk, fox, coyote, and badgers were not observed at scent-stations, operating scent-stations over the entire nesting season and through different habitats (e.g. wetland and woodland edges) would possibly have rendered a more accurate account of predator presence and activity.

Much concern has been generated regarding nesting success levels in the PPR. Reviewing long term data, Beauchamp et al. (1996) determined nest success had decreased since the 1930's with current levels below the 15% threshold required to maintain stable populations (Klett et al. 1988). Low mean nest success from the current study supports this general conception and is partially explained by a complex suite of parameters encompassing vegetation characteristics, habitat loss, alternative prey, and

predator community. Cover management may have indirect effects on nest success because management impacts vegetation characteristics that appear to have primary influence on small mammal and beetle abundance

The greatest alternative prey abundance occurred in YPM in which nest success was at its lowest, contrary to the Alternative Prey Hypothesis. Increased vole abundance might buffer some nests from depredation but because success was inversely related to beetle abundance, nests in habitats with more beetles appeared to be at greater risk. We often do not know predator composition and because the role of nesting cover patches in predator biology – considering the whole predator community – has not been fully explored, we can not directly determine the effect of alternative prey on nest success. Ackerman (2002) found no correlation between skunk activity and nest success and our predator activity indexes failed to produce usable correlations, yet both studies found a positive relationship between prey abundance and nest success. Future alternative prey-nesting studies within the PPR should address predator composition and patch use in conjunction with prey availability in a variety of landscapes. As predator communities tend to vary temporally and spatially, the use of permanent cover may change given vegetation and prey abundance. The importance of permanent cover to ducks is well known, but further knowledge of how other prairie inhabitants respond to patches of cover and the resulting interspecific relationships will enhance understanding of the dynamics of nest depredation.

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Table 3.1. Number of small mammals caught, total counts per species and year, and number of trap nights operated in July of 2000 and 2001 in the Aspen Parkland of Saskatchewan and Manitoba.

Species	Number Trapped		
	2000	2001	Total
Meadow vole *(Microtus pennsylvanicus)*	633	298	921
Deer mouse *(Peromyscus maniculatus)*	155	59	209
Masked shrew / Hayden's shrew *(Sorex cinereus/ haydeni)*	160	27	187
13-lined ground squirrel *(Spermophilus tridecemlineatus)*	26	35	61
Arctic shrew *(Sorex arcticus)*	32	5	38
Northern pocket gopher *(Thomomys talpoides)*	8	20	28
Red-backed vole *(Clethrionomys gapperi)*	10	5	15
Northern short-tailed shrew *(Blarina brevicauda)*	9	3	12
Jumping mouse *(Zapus spp.)*	3	8	11
unknown	1	2	3
Olive-backed pocket mouse *(Perognathus fasciatus)*	1	0	1
Total	1038	462	1500
Trap Nights (TN)	10,000	10,250	20,250

Table 3.2. Number of beetles caught, total counts per Family and year, and number of trap nights during pitfall trapping between May and July in 2000 and 2001 in the Aspen Parkland of Saskatchewan and Manitoba.

Family	Number Trapped		
	2000	2001	Total
Silphidae (Carrion beetle)	13,590	14,202	27,792
Carabidae (Ground beetle)	9,757	10,271	20,028
Curculionidae (Snout beetle)	1,281	665	1,946
Elateridae (Click beetle)	587	1,353	1,940
Cicindellidae (Tiger beetle)	353	442	795
Scarabidae (Dung beetle)	287	359	646
Tenebrionidae (Darkling beetle)	89	64	153
Histeridae (Hister beetle)	41	89	130
Byrrhidae (Pill beetle)	49	30	79
Dytiscidae (Predacious diving beetle)	15	7	22
Meliodae (Blister beetle)	8	13	21
Coccinellidae (Lady-bird beetle)	5	6	11
Hydrophilidae (Water scavenger beetle)	1	1	2
Total	26,063	27,502	53,565
Trap Nights (TN)	15,698	13,000	28,698

Table 3.3. Number of avian predators observed per species during two road-side surveys in the Aspen Parkland of Saskatchewan and Manitoba during June and July of 2001.

Species	Number Observed		
	Survey 1	Survey 2	Total
American crow (*Corvus brachyrhynchos*)	190	251	441
Black-billed magpie (*Pica pica*)	60	64	124
Red-tailed hawk (*Buteo jamaicensis*)	32	46	78
Franklin's gull (*Larus pipixcan*)	9	64	73
Ring-billed gull (*Larus delawarensis*)	13	23	36
Gull (*Larus spp.*)	10	17	27
Hawk	5	12	17
Northern harrier (*Circus cyaneus*)	7	7	14
Common raven (*Corvus corax*)	2	4	6
Swainson's hawk (*Buteo swainsoni*)	1	2	3
Snowy owl (*Nyctea scandiaca*)	0	1	1
American kestrel (*Falco sparverius*)	0	1	1
Total	329	492	821
Kilometers travelled	496	496	992

Table 3.4. Ranking of *a priori* management, *a priori* vegetation, landscape, and top exploratory models relating covariates with avian predator activity. Models are ranked within each suite by Akaike's Information Criterion, adjusted for small sample size (ΔAIC_c), and model weight ratio (w/w_i). Competing exploratory models were selected as those models with $\Delta AIC_c \leq 2$.

Suite and models	log (L)	K[a]	AIC_c	ΔAIC_c	w/w_i
A priori management suite					
NULL	29.0	2	33.32	0	1
A priori vegetation suite					
MXHT+MXHT^2	22.5	4	31.64	0	1
MXHT	25.8	3	32.47	0.82	0.6624
DUF+MXHT	24.1	4	33.24	1.60	0.4493
Landscape covariate suite					
CROP	21.4	3	28.07	0	1
CROP+CROP^2	20.8	4	29.94	1.88	0.3914
Exploratory suite					
MXHT+MXHT^2+CROP+GEO	7.0	6	21.55	0	1
MXHT+CROP+GEO	11.8	5	23.56	2.02	0.3643
MXHT+MXHT^2+CROP	11.9	5	23.66	2.12	0.3466
MXHT+MXHT^2+CROP+CROP^2+GEO	6.8	7	24.30	2.75	0.2523

Note: Model covariates are described in Appendix E.

[a] Number of model parameters including independent variables, dependent variable, intercept, random factor, and residual variance.

Table 3.5. Total number of mammalian tracks observed per species at scent-stations in 350 station nights in Saskatchewan and Manitoba of July 2001.

Species	Number of nights detected
Red fox (*Vulpes vulpes*)	18
Coyote (*Canis latrans*)	15
Weasel (*Mustela spp.*)	14
Raccoon (*Procyon lotor*)	7
Badger (*Taxidea taxus*)	6
Mink (*Mustela vison*)	5
Skunk (*Mephitis mephitis*)	4
Unknown	4
Dog (*Canis familiaris*)	1
Cat (*Felis catus*)	1
Total visits	75

Table 3.6. Ranking of *a priori* management, *a priori* vegetation, landscape, and top exploratory models relating covariates with mammalian predator activity. Models are ranked within each suite by Akaike's Information Criterion, adjusted for small sample size (ΔAIC_c), and model weight ratio (w/w_i). Competing exploratory models were selected as those models with $\Delta AIC_c \leq 2$.

Suite and models	log (L)	K[a]	AIC_c	ΔAIC_c	w/w_i
A priori management suite					
TRT	533.00	5	544.76	0	1
A priori vegetation suite					
MXHT+FORB	533.40	4	542.54	0	1
VOR+FORB	536.40	4	545.54	3.0	0.2231
Landscape covariate suite					
GEO	539.8	3	546.47	0	1
Exploratory suite					
MXHT+FORB+GEO	525.8	5	537.56	0	1

Note: Model covariates are described in Appendix E.
[a] Number of model parameters including independent variables, dependent variable, intercept, random factor, and residual variance.

Table 3.7. Species and number of upland duck nests monitored in planted cover in the Aspen Parkland of Saskatchewan and Manitoba during May through July 2000 – 01.

Species	Number of nests monitored		
	2000	2001	Total
Blue-winged teal (*Anas discors*)	231	222	453
Mallard (*Anas platyrhynchos*)	76	82	158
Northern shoveler (*Anas clypeata*)	97	57	154
Gadwall (*Anas strepera*)	55	31	86
American widgeon (*Anas americana*)	4	4	8
Green-winged teal (*Anas crecca*)	4	2	6
Northern pintail (*Anas acuta*)	3	0	3
Lesser scaup (*Aythya affinis*)	1	2	3
Total	471	400	871

Table 3.8. Ranking of *a priori* management, *a priori* vegetation, landscape, and top exploratory models relating covariates and vole abundance with nest survival. Models are ranked within each suite by Akaike's Information Criterion (ΔAIC) and model weight ratio (w/w_i). Competing exploratory models were selected as those models with ΔAIC ≤ 2.

Models	log (*L*)	K	AIC	ΔAIC	w/w_i
A priori management suite					
NULL	2071.4	2	2075.4	0	1
YR	2071.4	3	2077.4	2.0	0.3679
A priori vegetation suite					
MXHT+MXHT^2	2063.0	4	2071.0	0	1
Landscape covariate suite					
WA+WA^2+VOLE	2059.0	5	2069.0	0	1
BEET	2063.3	3	2069.3	0.3	0.8607
WA+WA^2	2062.0	4	2070.0	1.0	0.6065
VOLE+VOLE^2+WA+WA^2	2058.2	6	2070.2	1.2	0.5488
VOLE+BEET	2062.8	4	2070.8	1.8	0.4066
Exploratory suite					
MXHT+MXHT^2+WA+WA^2+BEET+VOLE	2042.2	8	2058.2	0	1
MXHT+MXHT^2+WA+WA^2+BEET	2044.5	7	2058.5	0.3	0.8607
MXHT+MXHT^2+WA+WA^2+VOLE+VOLE^2 +BEET	2041.0	9	2059.0	0.8	0.6703
YR+MXHT+MXHT^2+WA+WA^2+BEET+VOLE	2041.9	9	2059.9	1.7	0.4274
YPM+MXHT+MXHT^2+WA+WA^2+BEET+VOLE	2034.0	13	2060.0	1.8	0.4066
MXHT+MXHT^2+WA+WA^2+BEET+BEET^2+VOLE	2042.1	9	2060.1	1.9	0.3867

Note: Model covariates are described in Appendix F.

[a] Number of model parameters including independent variables, dependent variable, intercept, random factor, and residual variance.

Table 3.9. Ranking of *a priori* management, *a priori* vegetation, landscape, and top exploratory models relating covariates with nest survival 2001 data. Models are ranked within each suite by Akaike's Information Criterion (ΔAIC) and model weight ratio (w/w_t). Competing exploratory models were selected as those models with $\Delta AIC \leq 2$.

Models	log (*L*)	K	AIC_c	ΔAIC_c	w/w_t
A priori management suite					
NULL	2060.5	2	2064.5	0	1
A priori vegetation suite					
NULL	938.2	2	942.2	0	1
MXHT	937.5	3	943.5	1.3	0.5220
DUFF	937.6	3	943.6	1.4	0.4966
MXHT+MXHT^2	935.7	4	943.7	1.5	0.4724
FORB	938.1	3	944.1	1.9	0.3867
VOR	938.2	3	944.2	2.0	0.3679
Landscape covariate suite					
WA+WA^2	932.1	4	940.1	0	1
VOLE+WA+WA^2	932.1	5	942.1	2.0	0.3679
Exploratory suite					
BEET+AVE+WA+WA^2+MXHT+MXHT^2	922.6	8	938.6	0	1
AVE+DNC+DNC^2	928.8	5	938.8	0.2	0.9048
BEET+AVE+DNC+DNC^2+MXHT+MXHT^2	922.9	8	938.9	0.3	0.8607
BEET+AVE+MXHT+MXHT^2	927.4	6	939.4	0.8	0.6703
BEET+AVE+WA+WA^2+MXHT	925.5	7	939.5	0.9	0.6376
BEET+AVE+DNC+MXHT	927.7	6	939.7	1.1	0.5769
BEET+AVE+WA+MXHT	927.7	6	939.7	1.1	0.5769
AVE+DNC	931.9	4	939.9	1.3	0.5220
AVE+WA+WA^2	929.9	5	939.9	1.3	0.5220
BEET+DNC+MXHT+MXHT^2	928	6	940.0	1.4	0.4966
BEET+WA+WA^2+MXHT+MXHT^2	926.2	7	940.2	1.6	0.4493
BEET+AVE+DNC+DNC^2+MXHT	926.2	7	940.2	1.6	0.4493
AVE+WA+WA^2+MXHT	928.3	6	940.3	1.7	0.4274
BEET+AVE+MXHT	930.3	5	940.3	1.7	0.4274
AVE+WA+WA^2	930.5	5	940.5	1.9	0.3867

Note: Model covariates are described in Appendix F.
[a] Number of model parameters including independent variables, dependent variable, intercept, random factor, and residual variance.

Figure 3.1. Ecoregion boundaries and locations of study sites sampled during 2000 and 2001 in Saskatchewan and Manitoba. Study sites are symbol coded to treatment type.

Figure 3.2. Schematic diagram of study design showing advancement of projects in stand age post-management (YPM) between sampling years 2000 and 2001.

Figure 3.3. Diagram of the 5 by 5 grid layout used in small mammal sampling. Two Museum Special snap-traps were placed at each grid point with 10 m spacing between grid points.

Figure 3.4. Diagram of a pitfall trap used for insect sampling and placement of the trap at ground level. The trap consists of an outer plastic cup and removable inner liner filled with a saturated salt solution.

Figure 3.5. Relationship between maximum height and avian predator activity (ln[x+1] of # observed/km) for projects sampled in 2001.

Figure 3.6. Relationship between percent cropped lands (arcsine transformed) and avian predator activity (ln[x+1] of # observed/km) ±95% CI for projects sampled in 2001

Figure 3.7. Relationship between geographic index and avian predator activity (ln[x+1] of # observed/km) ±95% CI for projects sampled in 2001. Geographic index is derived

from the product of degrees latitude and longitude where index values increase in a north-westwardly direction.

Figure 3.8. Mean mammalian predator activity ±95% CI per treatment for projects sampled in July of 2001.

Figure 3.9. Relationship between percent forb composition (arcsine transformed) and mammalian predator activity (# present per 1000 station nights) ±95% CI for projects sampled in July 2001.

Figure 3.10. Relationship between maximum height (dm) and mammalian predator activity (# present per 1000 station nights) ±95% CI for projects sampled in July 2001.

Figure 3.11. Relationship between geographic index and mammalian predator activity (# present per 1000 station nights) ±95% CI for projects sampled in July 2001. Geographic index is derived from the product of degrees latitude and longitude where index values increase in a northwestwardly direction.

Figure 3.12. Nest survival estimates (Mayfield extension ±1 SE) per treatment and year post-management for upland duck nests monitored in Manitoba and Saskatchewan in 2000 and 2001.

Figure 3.13. Estimated nest survival (Mayfield extension) in relation to mean maximum vegetation height for permanent cover projects sampled in Saskatchewan and Manitoba in 2000 and 2001. Estimates are based on model $y=8.5405-2.0766x+0.1813x^2$.

Figure 3.14. Estimated nest survival (Mayfield extension) in relation to the ratio of wetland area to permanent cover (DNC) for projects sampled in Saskatchewan and Manitoba in 2000 and 2001. Estimates are based on model $y=1.6702+2.1279x-0.8467x^2$ where the wetland area has been log transformed.

Figure 3.15. Estimated nest survival (Mayfield extension) in relation to vole abundance for permanent cover projects sampled in Saskatchewan and Manitoba in 2000 and 2001. Estimates are based on model $y=2.595+0.108x$ where vole abundance has been log transformed.

Figure 3.16. Estimated nest survival (Mayfield extension) in relation to beetle abundance for permanent cover projects sampled in Saskatchewan and Manitoba in 2000 and 2001. Estimates are based on model $y=3.9924-0.2883x$ where beetle abundance has been log transformed.

Figure 3.17. Estimated nest survival (Mayfield extension) in relation to avian predator activity for projects sampled in Saskatchewan and Manitoba in 2000 and 2001. Estimates are based on model $y=2.4467+0.4499x^2$ where predator activity has been log transformed.

Figure 3.18. Estimated nest survival (Mayfield extension) in relation to project size (amount of DNC) for projects sampled in Saskatchewan and Manitoba in 2000 and 2001. Estimates are based on model $y=2.4137+0.03576x-0.00067x^2$.

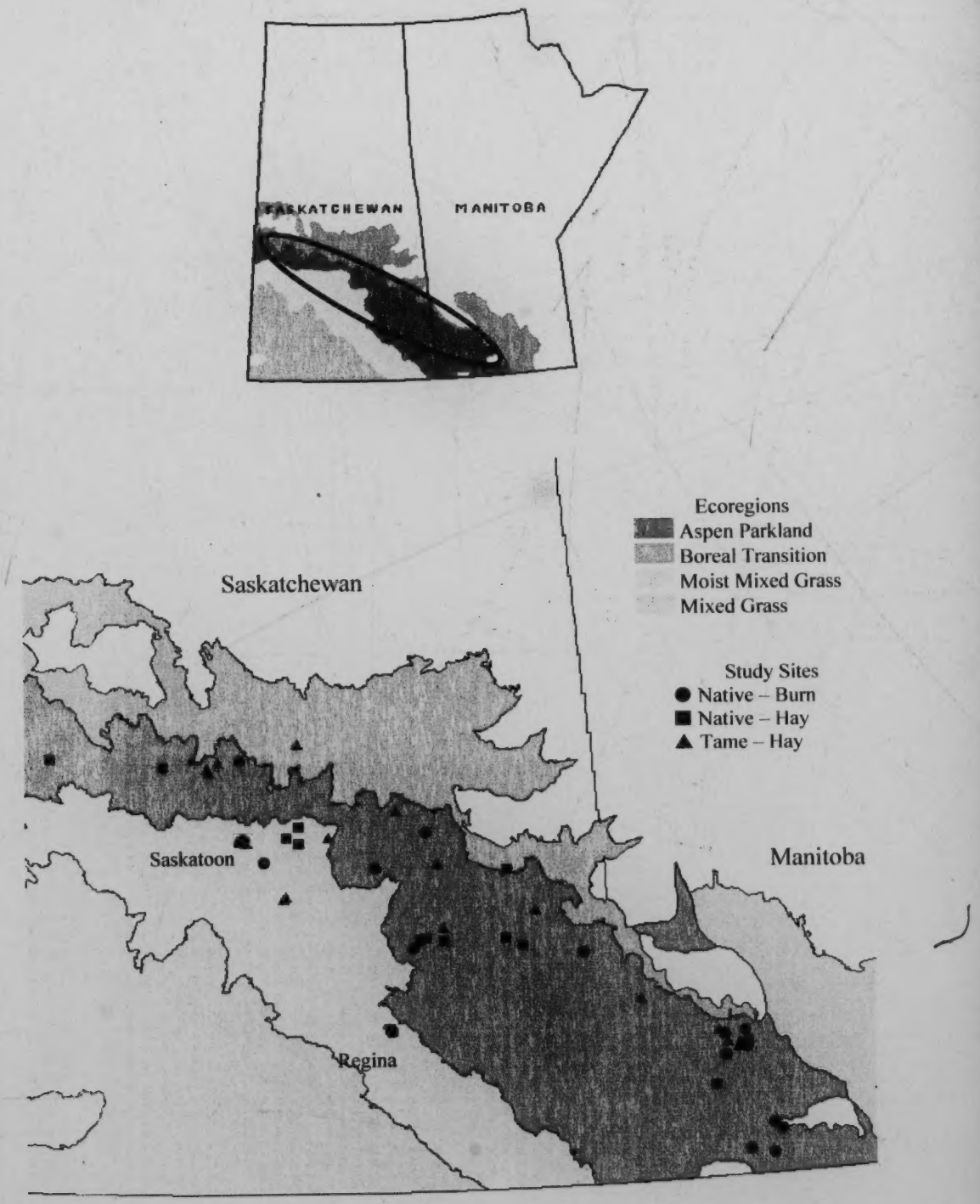
SASKATCHEWAN
MANITOBA
Ecoregions
Aspen Parkland
Boreal Transition
Moist Mixed Grass
Mixed Grass
Study Sites
Native – Burn
Native – Hay
Tame – Hay
Saskatchewan
Saskatoon
Manitoba
Regina

Figure 3.1

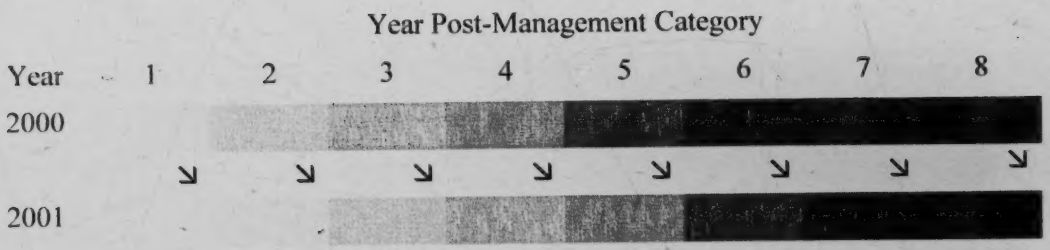

Figure 3.2

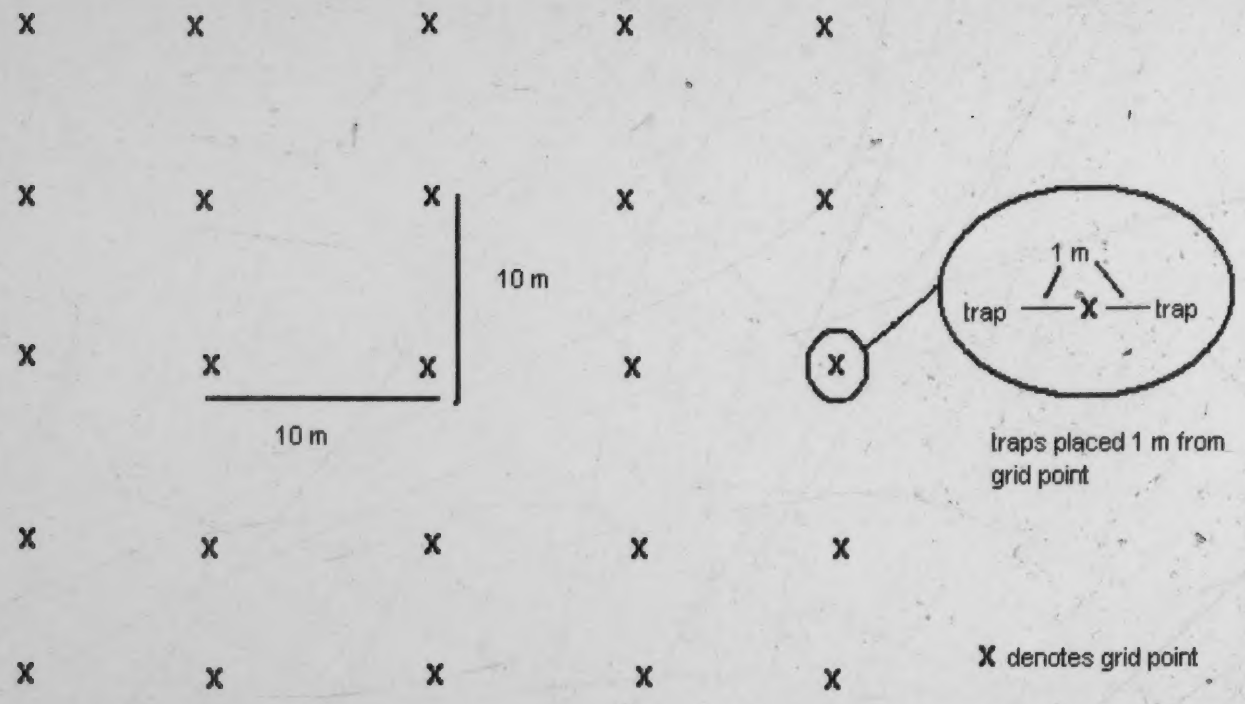
10 m
10 m
1 m
trap
trap
traps placed 1 m from grid point
X denotes grid point

Figure 3.3

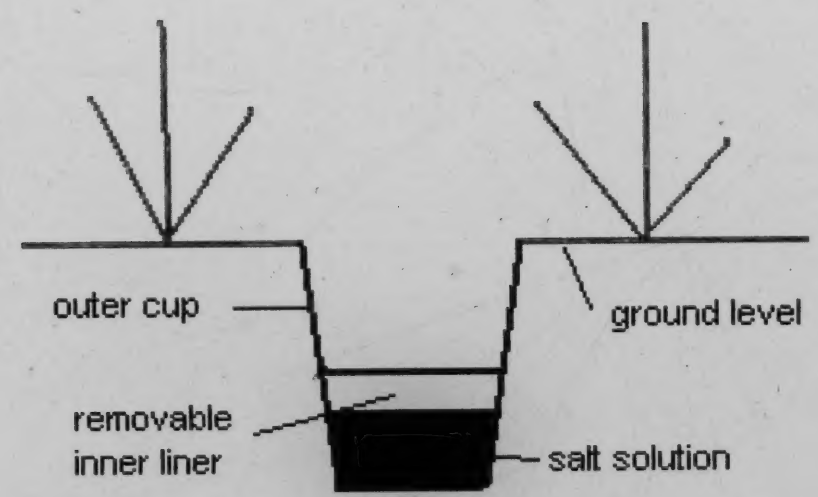
outer cup
ground level
removable
inner liner
salt solution

Figure 3.4

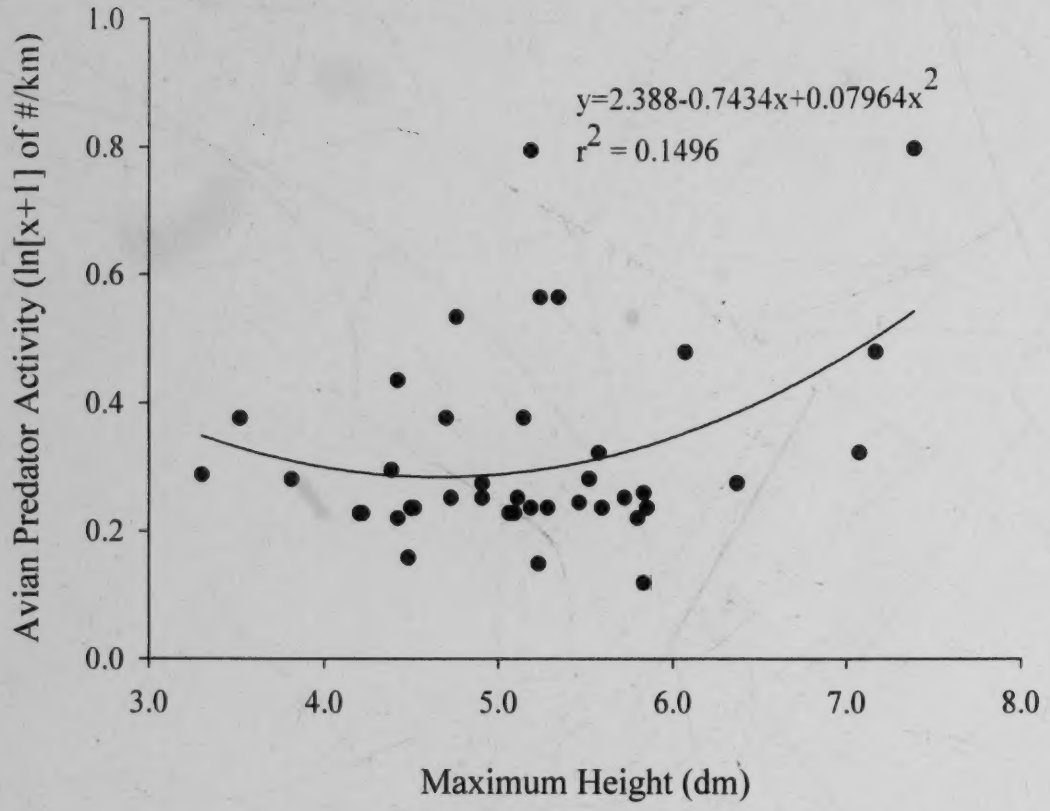
y=2.388-0.7434x+0.07964x2
r2 = 0.1496
Avian Predator Activity (ln[x+1] of #/km)
Maximum Height (dm)
1.0
0.8
0.6
0.4
0.2
0.0
3.0
4.0
5.0
6.0
7.0
8.0

Figure 3.5

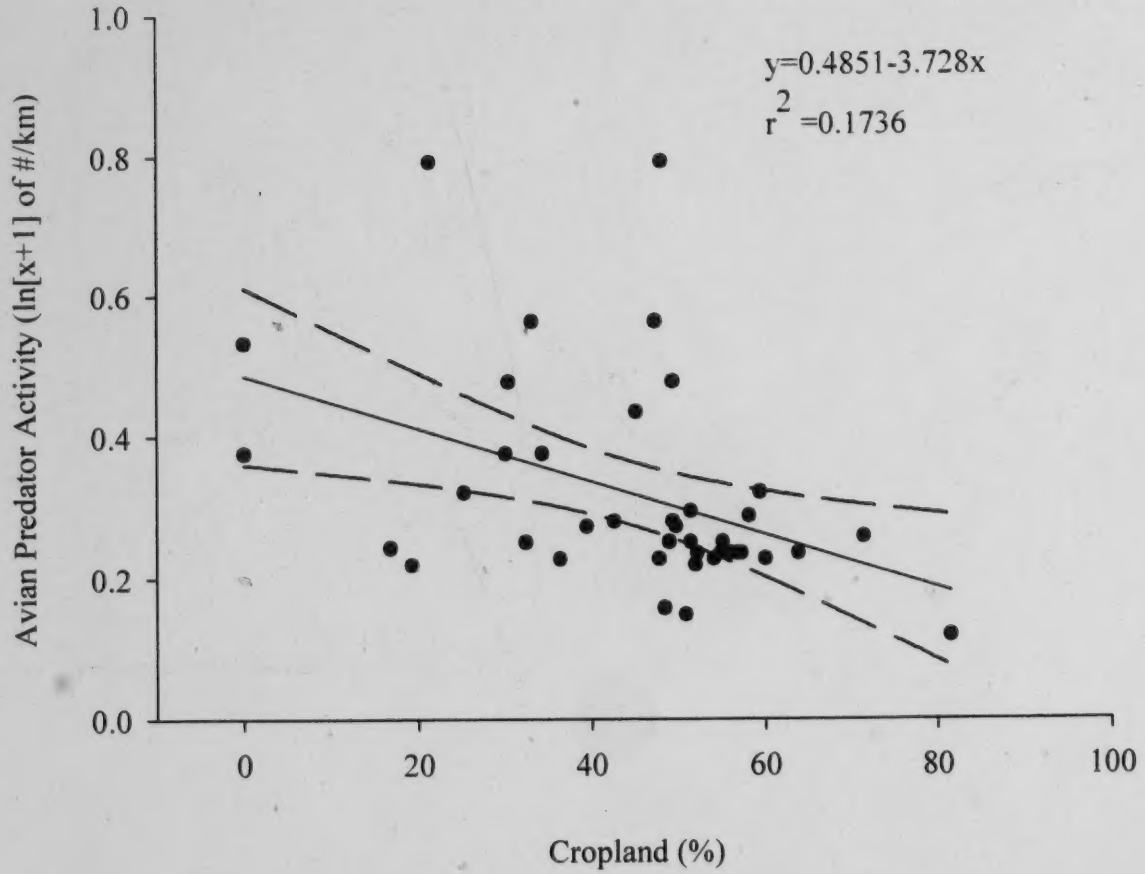
y=0.4851-3.728x
r 2 =0.1736
Avian Predator Activity (ln[x+1] of #/km)
1.0
0.8
0.6
0.4
0.2
0.0
0
20
40
60
80
100
Cropland (%)

Figure 3.6

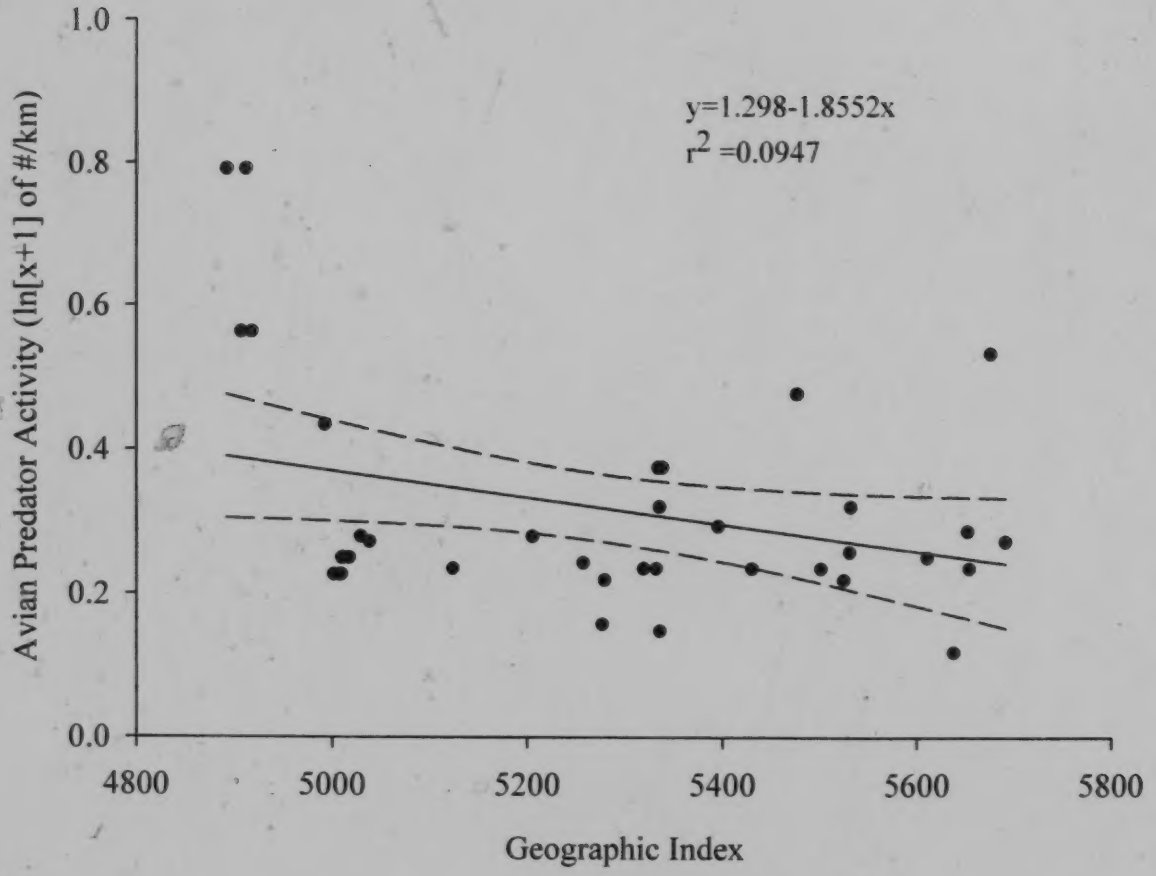
y=1.298-1.8552x
r2 =0.0947
Avian Predator Activity (ln[x+1] of #/km)
1.0
0.8
0.6
0.4
0.2
0.0
4800
5000
5200
5400
5600
5800
Geographic Index

Figure 3.7

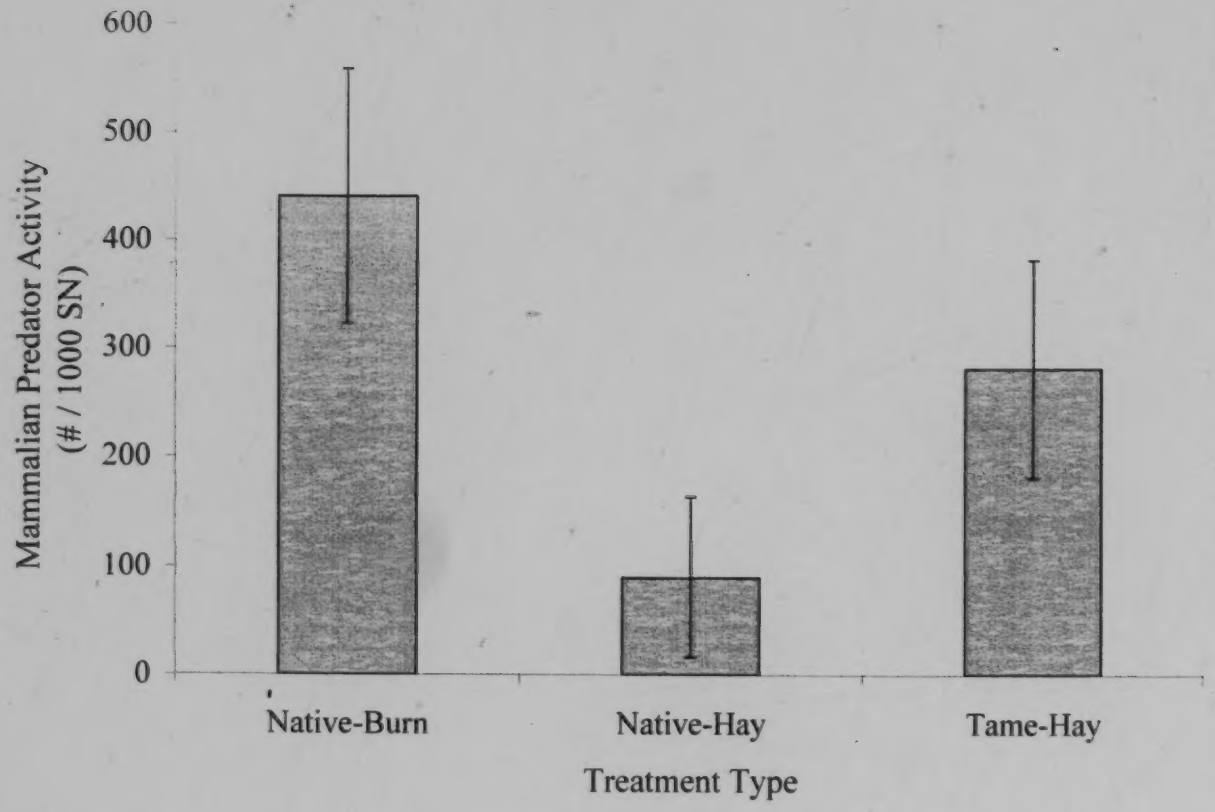
600
500
400
300
200
100
0
Mammalian Predator Activity
(# / 1000 SN)
Native-Burn
Native-Hay
Tame-Hay
Treatment Type

Figure 3.8

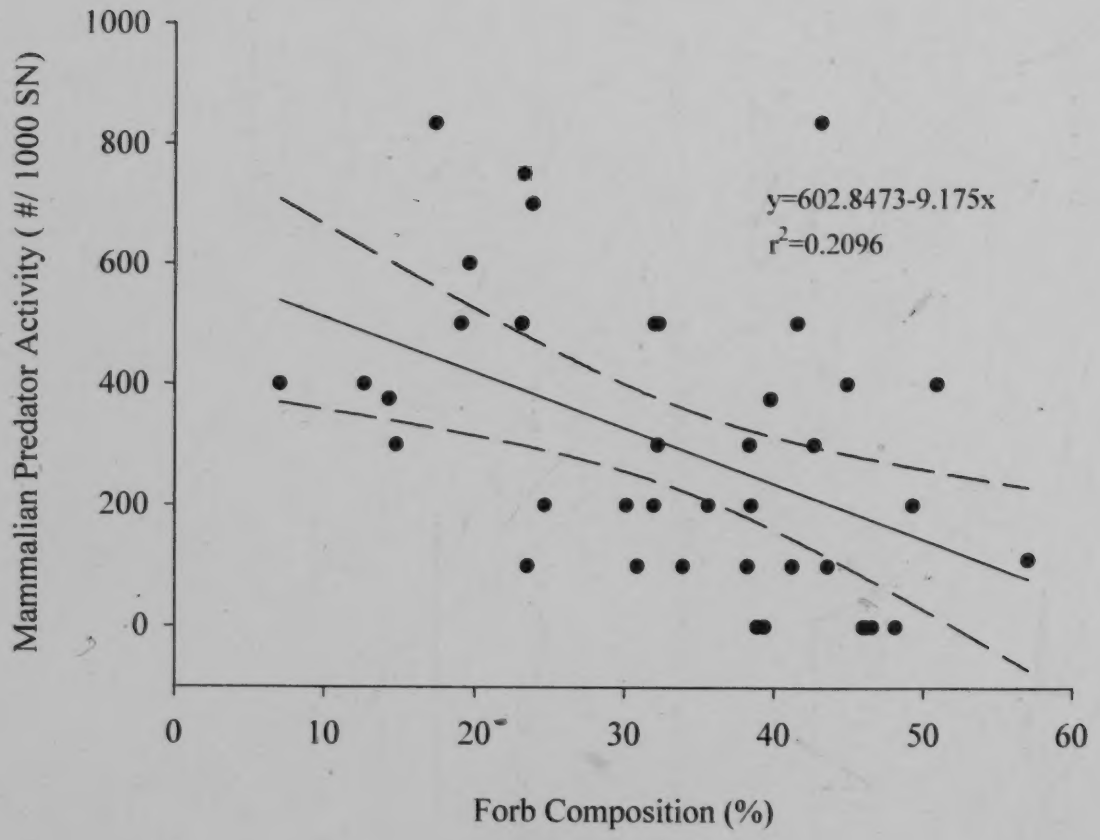
Mammalian Predator Activity (#/ 1000 SN)
1000
800
600
400
200
0
0
10
20
30
40
50
60
Forb Composition (%)
y=602.8473-9.175x
r2=0.2096

Figure 3.9

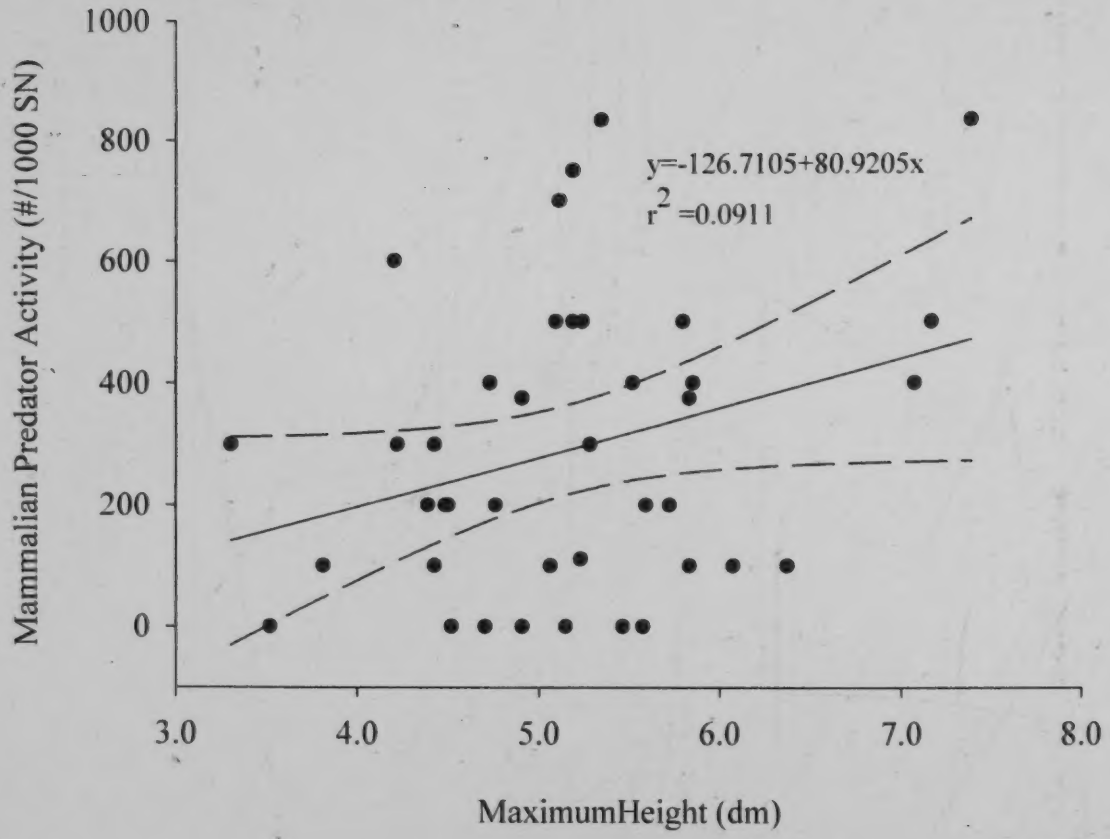
Mammalian Predator Activity (#/1000 SN)
1000
800
600
400
200
0
y=-126.7105+80.9205x
r2 =0.0911
3.0
4.0
5.0
6.0
7.0
8.0
MaximumHeight (dm)

Figure 3.10

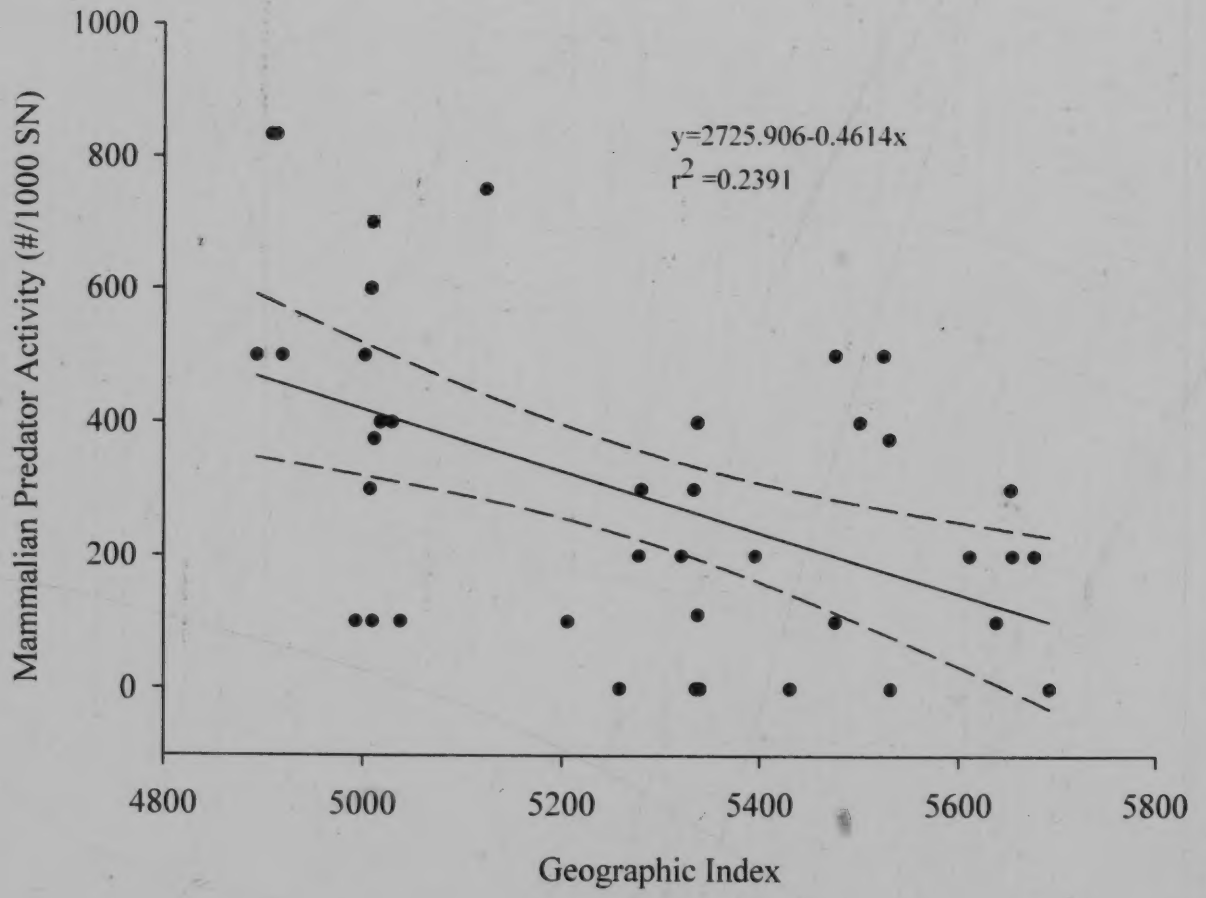
y=2725.906-0.4614x
r2 =0.2391
Mammalian Predator Activity (#/1000 SN)
1000
800
600
400
200
0
4800
5000
5200
5400
5600
5800
Geographic Index

Figure 3.11

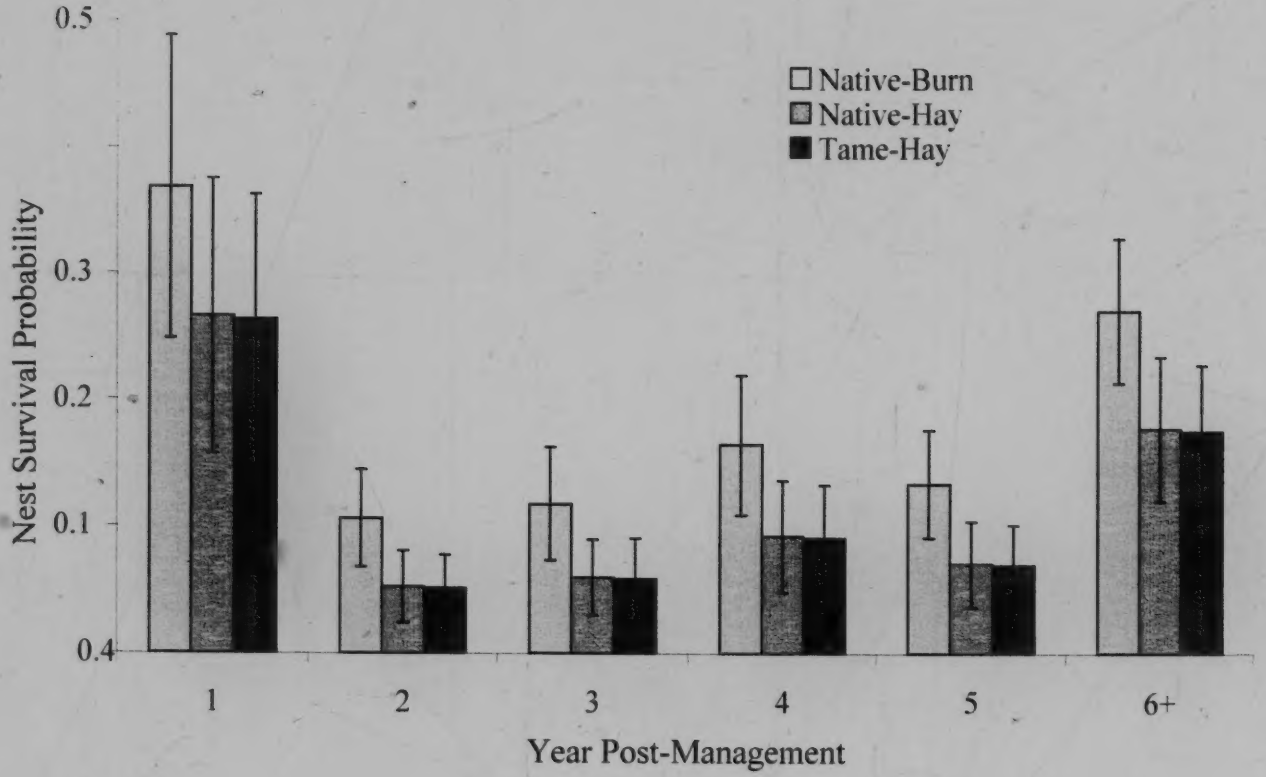
0.5
0.3
0.2
0.1
0.4
Nest Survival Probability
Native-Burn
Native-Hay
Tame-Hay
1
2
3
4
5
6+
Year Post-Management

Figure 3.12

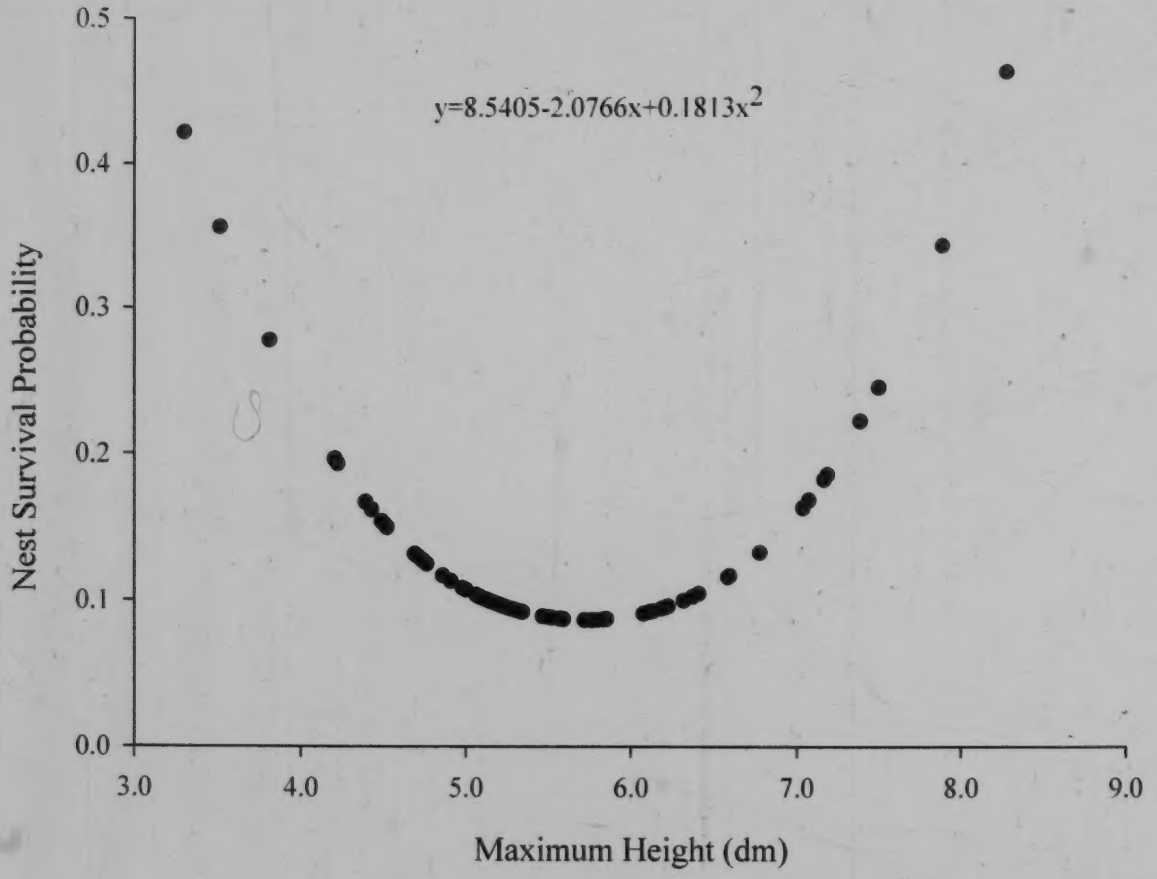
y=8.5405-2.0766x+0.1813x2
Nest Survival Probability
0.5
0.4
0.3
0.2
0.1
0.0
3.0
4.0
5.0
6.0
7.0
8.0
9.0
Maximum Height (dm)

Figure 3.13

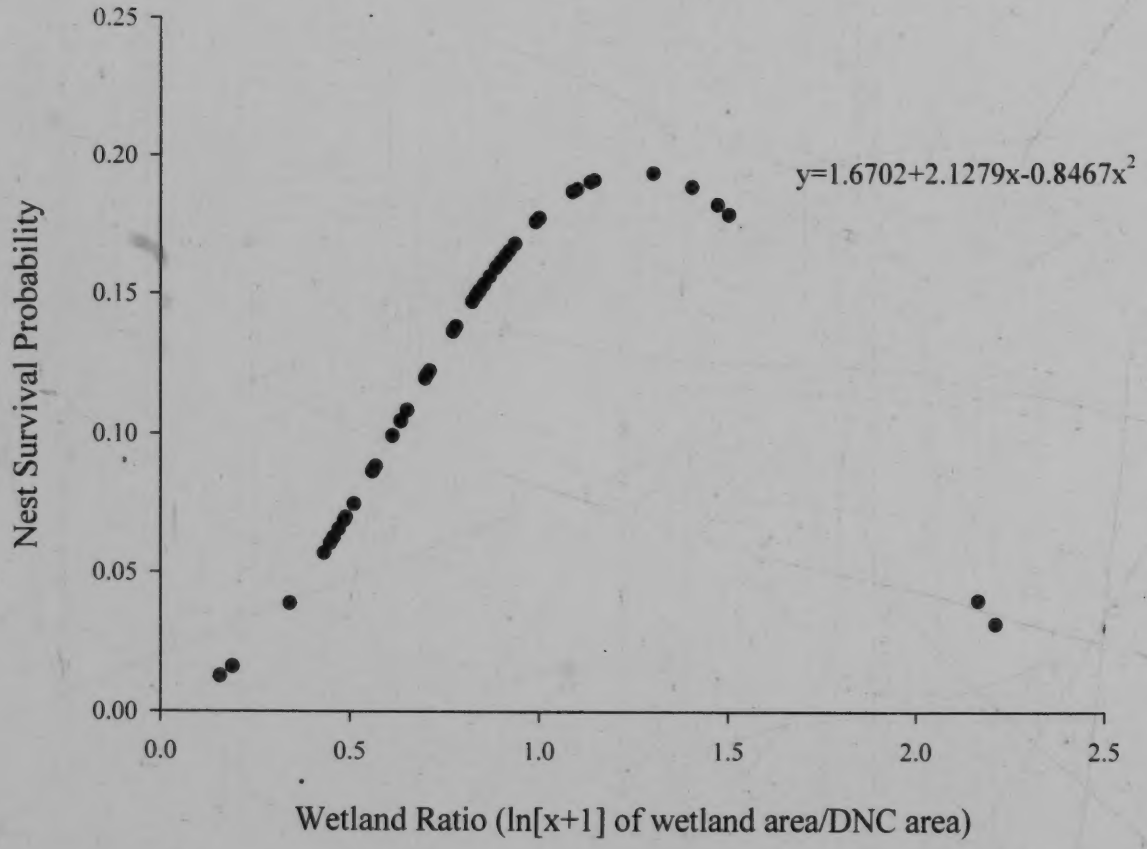
0.25
0.20
0.15
0.10
0.05
0.00
Nest Survival Probability
y=1.6702+2.1279x-0.8467x²
0.0
0.5
1.0
1.5
2.0
2.5
Wetland Ratio (ln[x+1] of wetland area/DNC area)

Figure 3.14

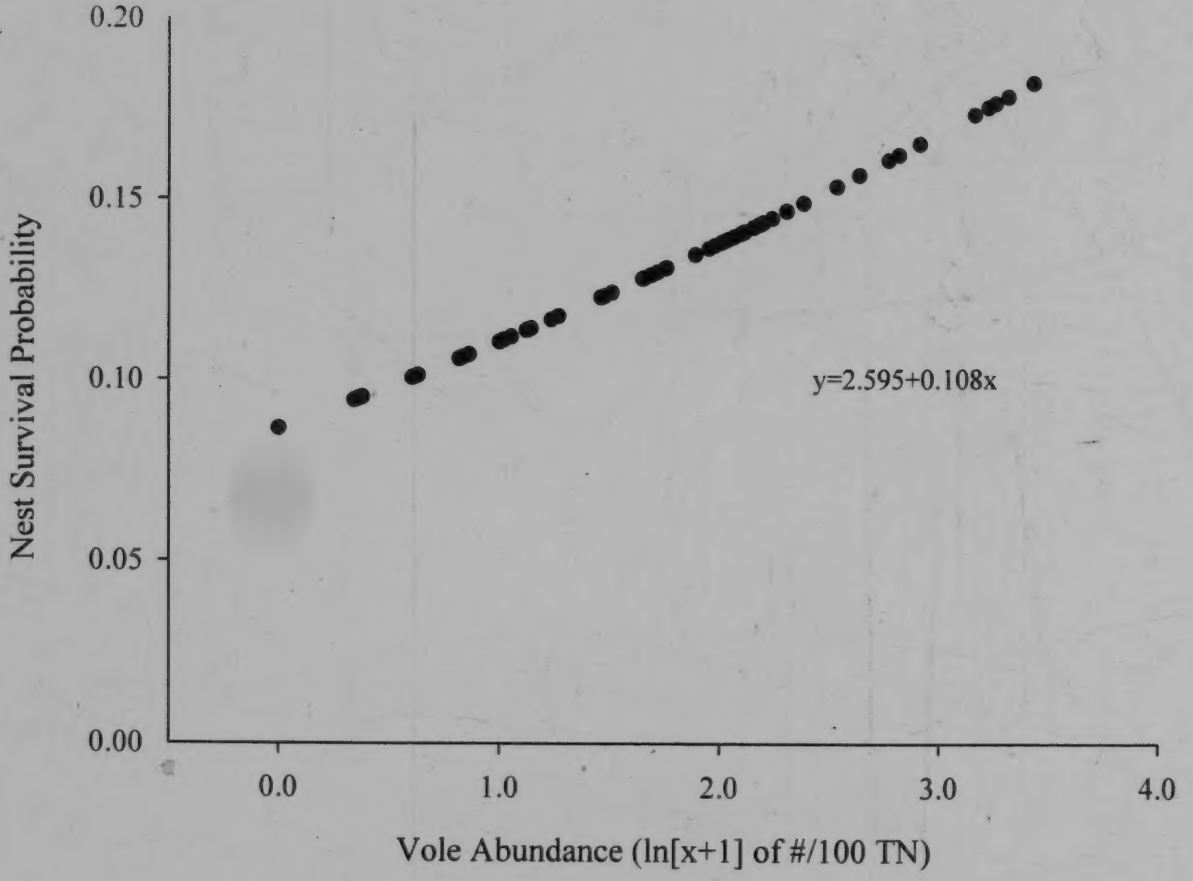
0.20
0.15
0.10
0.05
0.00
Nest Survival Probability
y=2.595+0.108x
0.0
1.0
2.0
3.0
4.0
Vole Abundance (ln[x+1] of #/100 TN)

Figure 3.15

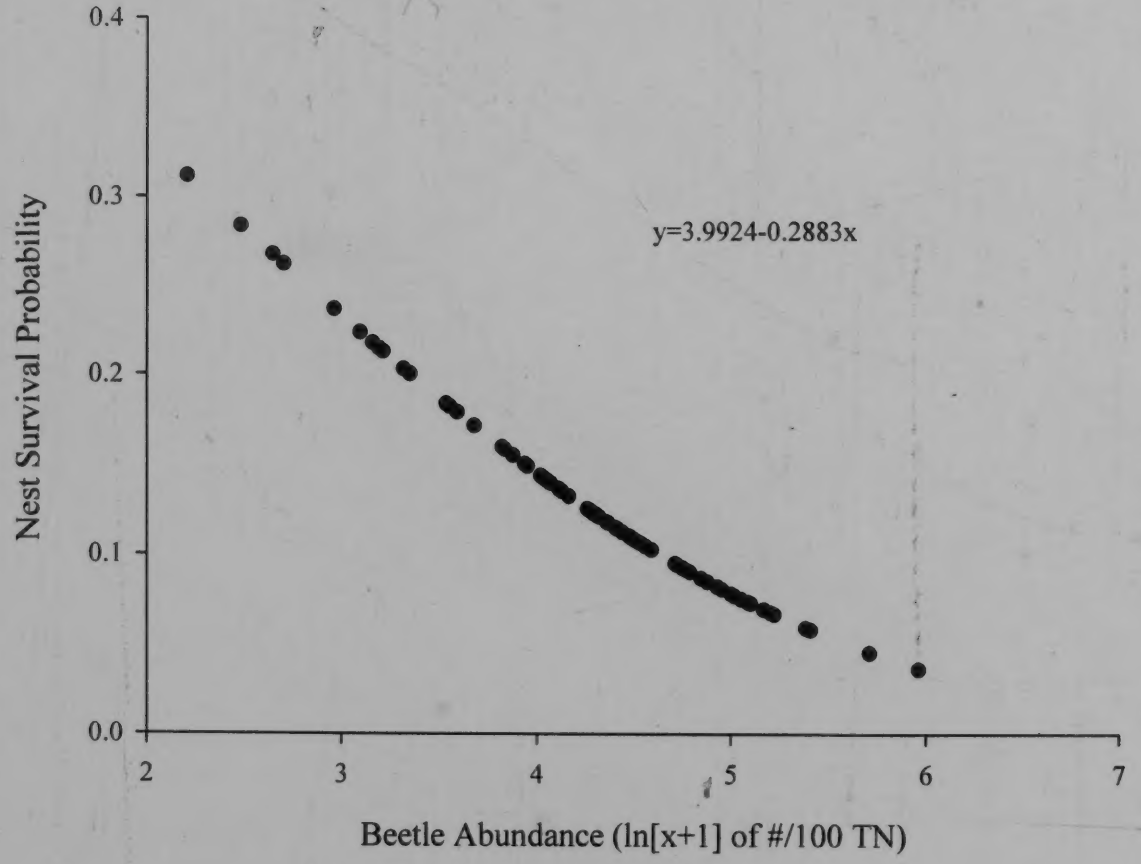
0.4
0.3
0.2
0.1
0.0
Nest Survival Probability
y=3.9924-0.2883x
2
3
4
5
6
7
Beetle Abundance (ln[x+1] of #/100 TN)

Figure 3.16

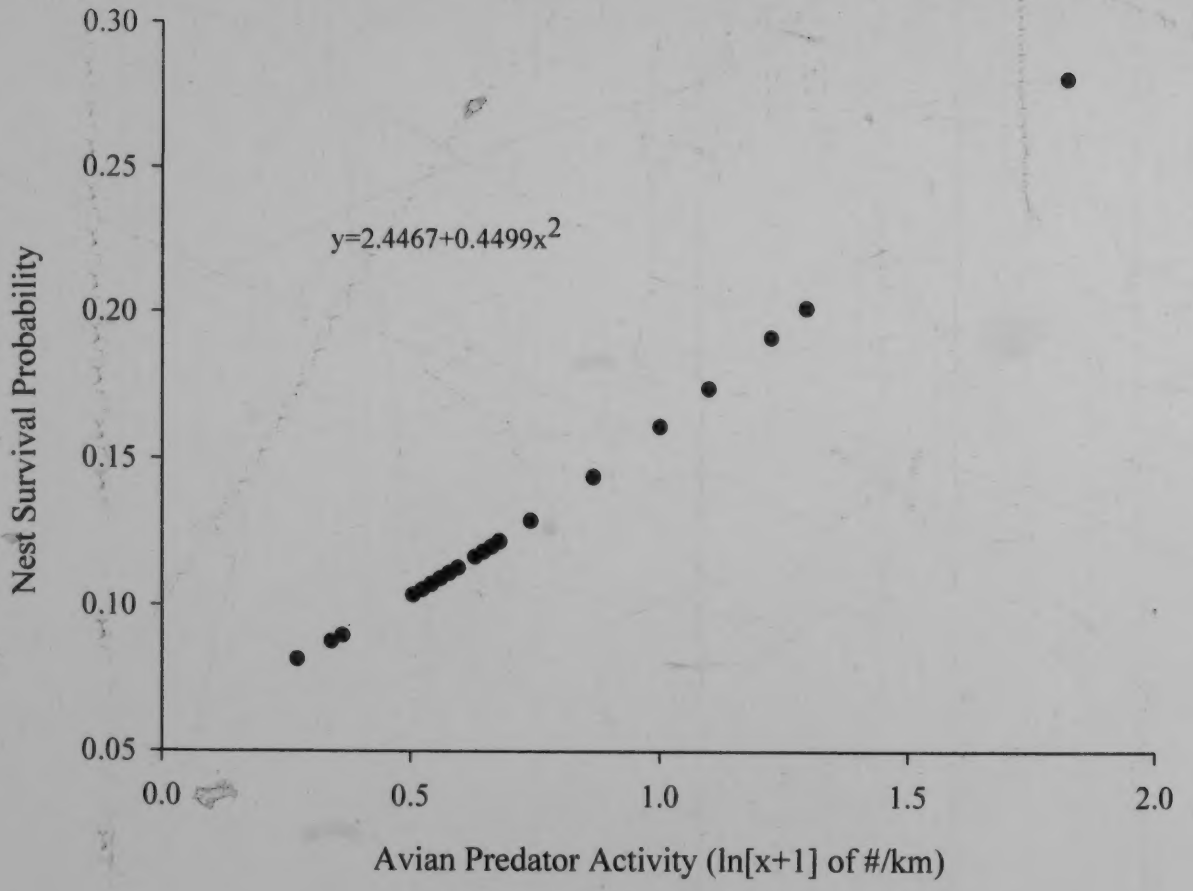
0.30
0.25
0.20
0.15
0.10
0.05
Nest Survival Probability
y=2.4467+0.4499x2
0.0
0.5
1.0
1.5
2.0
Avian Predator Activity (ln[x+1] of #/km)

Figure 3.17

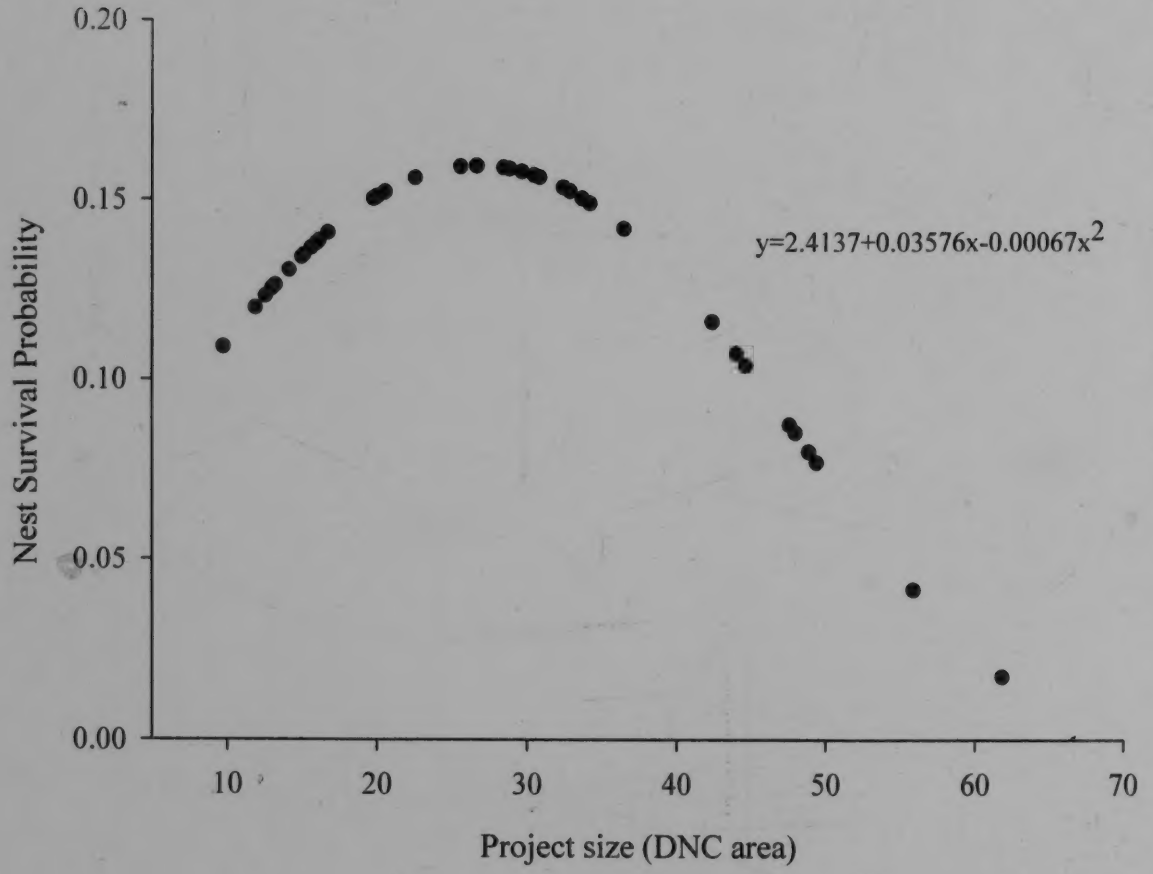
0.20
0.15
0.10
0.05
0.00
Nest Survival Probability
y=2.4137+0.03576x-0.00067x2
10
20
30
40
50
60
70
Project size (DNC area)

Figure 3.18

CHAPTER 4

GENERAL CONCLUSIONS

General Discussion

Abundance of alternative prey items has been shown to buffer depredation on grouse, duck, and goose nests but few studies have concentrated on this relationship within managed cover in the PPR of Canada. Even though skunks are a principal nest predator and depredation on ground nests is largely considered incidental, this is the first study to consider beetles as alternative prey items. The first step in examining the role of alternative prey was to determine prey abundance in managed waterfowl nesting cover.

The models generated show management techniques, for the most part, were not driving prey abundance or nest success. Rather, vegetation parameters were the main determinants for small mammal abundance. Deer mice were the exception to this where abundance peaked after a management event. Still, the total number of small mammals captured increased linearly with vegetation density. Beetle models, on the other hand, were best modelled with duff depth and conserved soil moisture. Stand age was important for Carabid abundance with greater captures in YPM 2 and 3. Overall, beetle and Carabid abundance decreased with greater duff depth and conserved soil moisture.

Vegetation height, wetland area, and alternative prey abundance were key in determining nest success. However, the relationship between prey abundance and nest success was contrary as to what was hypothesized under the alternative prey hypothesis (Angelstam et al. 1984). While nest success increased slightly with greater vole abundance, it plummeted as beetle abundance increased. Carabid abundance and small mammal abundance were greatest in YPM 2 through 4, which coincided with the lowest

nest success rates. Therefore, it is fair to assume that under these conditions, more alternative prey, in the form of beetles, may result in severely limited waterfowl recruitment.

Study limitations and management implications

An important limitation to this study was the inability to accurately describe the predator community and activity levels and relate it to nest success during the nest search intervals. Furthermore, we require small mammal abundance data during the same search intervals. Predators may focus on different prey items at different times during the summer and being able to directly relate predation pressure given prey abundance, nest success, and habitat characteristics would provide a better idea of where management should be directed. If we assume nest success was higher for early nesters (May nests) than later nesters (June – July) as has been shown by Howerter (2003), then alternative prey abundance is most critical for early nest success. Greater success of May nests is also imperative since the majority of recruitment is generated through early nesters (Blums et al. 2002). Predators are prey limited during this period and so should be encountering nests more frequently, due to increased search time. However, predators may be avoiding dense cover early in the season, opting for sparse cover and edges that are more easily searched.

Nest survival models suggest vegetation height and more wetland area relative to upland area – to a point – results in higher nest success. Wetland density has often been attributed to nest success. Skunks and raccoons (Greenwood 1982; Phillips et al. 2003) tend to focus on wetland edges. Where few edges exist, prey may be exhausted quickly forcing predators to forage in uplands and in direct association with nests. Where

medium amounts of edge exist, predators may take more time to exhaust resources and thus, less likely to move into uplands. Nests in close proximity to wetlands suffer higher depredation rates (Howerter 2003), therefore where the wetland to upland ratio is large, nests, by default, are closer to edges and easily encountered. In areas with high skunk concentrations or where skunks are the main predator, it may be beneficial to determine how beetle abundance differs within habitats (i.e. abundance near wetlands versus nesting habitat core).

During this study, we witnessed a decrease of 50% in small mammal abundance between years yet nest success remained constant. This draws to light the unpredictability of rodent abundance and while voles may have buffered predation one year, this is most likely not ubiquitous across the prairies and over time. However, stands require management to remain healthy (Naugle et al. 2000) and because small mammals were more abundant the first 4 years post-management, managing dense cover every 4 or 5 years may maintain elevated rodent densities and quality nesting cover. While not directly managing to sustain high rodent densities, the result may be a positive spin-off. A positive correlation between vole abundance and nest success existed in California (Ackerman 2002) and voles seem to have a small role in buffering predation in this study. But whether elevated densities will persist over time – under managed conditions – needs to be investigated further. On the other hand, elevated rodent densities may work to attract predators, thus the patch may become a sink. This is entirely plausible if we consider predators capable of recognizing profitable habitats. In this instance, cover management will not serve to mitigate nest depredation.

Contrary to vole abundance, beetle abundance increased between years. Other than Carabid abundance, stand management apparently had little influence on overall beetle abundance. Perhaps more detailed data on microsite conditions (e.g. soil type, temperature, pH, and moisture) linked with species presence at traps would promote greater understanding of beetle communities in permanent cover. Further research should be directed at the intricacies of beetle population dynamics within managed cover and grassland habitats, especially if future research supports the inverse alternative prey relationship between beetles and nest success. A meta-analysis of published and unpublished small mammal and invertebrate data may provide insight into prairie prey populations and would be an important step in addressing gaps in existing knowledge.

Management alternatives such as providing supplemental food, conditioned taste aversion, and establishing predator barriers may increase nest success but do nothing to affect predation once the brood reaches water. Another alternative, predator removal, has been shown to be ineffective, especially if a multi-species predator community is present (Beauchamp et al. 1996) and removal of all predators, mammalian and avian, would be incredibly cost prohibitive, even at local scales. Short-term predator programs are often inadequate (Dion et al. 1999) and feasibility of long-term programs is subject to budgetary constraints. Unknown trophic effects and growing public concern regarding humane removal methods and the ethics of predator control, especially when native predators are involved, are further deterrents. Recent lawsuits brought forth by animal rights groups have put removal programs in jeopardy in the United States, even though management was directed at conserving endangered species (Goodrich and Buskirk 1995).

Large scale reversal of fragmentation and habitat loss in the PPR is implausible; management objectives must work within current limitations but with future goals in mind. Collaboration with government, local organizations, and community members, in addition to public education, are critical to success. Sovada et al. (2001) have suggested pro-coyote management would benefit waterfowl production by reducing meso-predator numbers. But attitudes toward predators, especially in an agricultural setting with livestock – be it cattle or game animals – are slow in changing. Concentrating on changing hunting pressures on large canid species could benefit ducks such that red fox numbers as well as skunks and raccoons are controlled by the presence of coyotes. Reduction of predator numbers, coupled with more effective government incentives aimed at crop diversification or returning marginal cropland to wildlife habitat could potentially increase duck recruitment.

Areas of high duck production tend to be associated with areas of high grassland composition (Kantrud 1993, Phillips et al. 2003), therefore, concentrating habitat programs in focal areas may be more economically feasible than small, isolated patches. Focal patches with adequate wetland density, sufficient nesting cover, and few nest predators need to be identified and recognized by wildlife/conservation organizations along with provincial/federal environmental departments. Preserving prairie wetlands has recently garnered much attention but for broods to make use of these wetlands, they must survive incubation. Prairie environments need to be protected but with the extent of fragmentation, protection will require innovative ideas and large-scale cooperation to achieve the desired result. Large-scale conservation agreements, incentives to plant wildlife friendly crops, or enhanced land stewardship require a considerable commitment

from all stakeholders. Because we do not know the full extent to which trophic levels are disrupted when native predators are removed for the benefit of a single species, the limited financial resources could be better spent on strategic habitat improvement than predator removal.

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Appendix A. Project characteristics which include project identifier, geographic location, amount of planted cover, year project was seeded, and treatment category (seed mix with number of years post-management) in 2000 and 2001.

Province	Project name & Number	Latitude	Longitude	Area (ha)[1]	Year Seeded[2]	2000 Category[3]	2001 Category
MB	Menzies	50.31000	-100.13305	14.1	1994	th3	th4
MB	Cooper	50.19980	-100.03211	11.4	1992	th6	th7
MB	Johnson	50.14749	-99.89416	33.6	1993	th4	th5
MB	McLeod	49.38027	-99.37722	26.6	1994	nb4-f	nb5-f
MB	Good	49.16639	-99.50916	28.4	1992	nb5-f	nb6-f
MB	Apache	49.81611	-100.21749	16.1	1992	nh7	nh8
MB	Lidcliff	50.63515	-101.17710	15.1	1985	th5	th6
MB	Myers	49.43266	-99.48954	15.9	1991	nb2-f	nb3-f
MB	Turner	49.21083	-99.81666	19.7	1991	nb2-f	nb3-f
MB	Kolesar	50.28027	-99.78583	42.4	1995	nb2-s	nb3-s
MB	Jackson	50.25833	-100.06333	20.5	1995	nb2-s	nb3-s
MB	Borotsik2	50.06305	-100.07138	9.7	1991	nb4-s	nb5-s
MB	Martens	50.06666	-100.08972	22.5	1994	nb4-f	nb5-f
MB	Scott	50.16500	-99.82999	26.7	1993	nb5-s	nb6-s
MB	Sharpe	50.18416	-99.76888	20.5	1993	nb5-f	nb6-f
MB	Harland	50.13305	-99.76944	13.1	1991	nb7-s	nb8-s
SK	Fontaine	52.26000	-105.84666	28.8	1990	nh4	nh5
SK	Valentine	51.88520	-104.78651	50.3	1992	nb4-s	-
SK	Kuhn	52.99527	-105.87138	16.0	1994	th3	th4
SK	Ponath	52.39456	-104.47052	5.6	1992	th8	-
SK	Ebner	52.16915	-105.43780	15.6	1992	th2	th3
SK	Thompson	52.79555	-107.80027	16.7	1993	nh6	nh7
SK	Schlichemeyer	52.84527	-106.68666	48.8	1992	nb5-s	nb6-s

Province	Project name & Number	Latitude	Longitude	Area (ha)[1]	Year Seeded[2]	2000 Category[3,4]	2001 Category
SK	Zbitnoff	52.83777	-107.00250	12.5	1993	th6	th7
SK	Bashuk	52.77916	-107.17194	40.0	1994	th2	-
SK	Saluk	52.85972	-107.37638	47.6	1998	-	th1
SK	Redberry	52.75092	-107.14690	47.9	1995	-	th1
SK	Meilicke	51.93972	-106.35722	11.8	1992	nb3-f	nb4-f
SK	Hanuschak	52.11098	-105.85369	9.1	1991	nh8	-
SK	Denis	52.16184	-106.02156	44.6	1992	nh3	nh4
SK	Miller	51.63805	-106.04611	36.4	1994	th1	th2
SK	Kuntz	51.62361	-106.06888	55.8	1994	th1	th2
SK	Jesmer	51.92083	-103.91361	14.9	1991	th4	th5
SK	Start	51.34653	-103.85141	44.0	1990	th5	th6
SK	Gyorfi	51.25138	-104.16444	61.8	1991	nh3	nh4
SK	Labash	51.25361	-104.09583	32.4	1991	nh5	nh6
SK	Bigoraj	51.84138	-102.93416	49.4	1993	nh2	nh3
SK	Lambert	51.22972	-103.83972	32.8	1992	nh2	nh3
SK	Orban	51.17194	-104.26888	30.4	1991	-	nb2-s
SK	Ingram	52.18305	-104.05944	25.6	1995	-	nb2-s
SK	Vokali	51.25166	-104.08277	29.6	1994	-	nh1
SK	Horseshoe	51.47140	-102.56618	12.9	1987	th7	th8
SK	Cursons	51.23596	-102.99538	30.8	1991	nh5	nh6
SK	Peterson	51.06638	-101.93138	34.2	1993	nb5-f	nb6-f
SK	White	51.16888	-102.75000	20	1994	nh4	nh5

Notes: [1]Acreage represents the digitized area of planted cover (i.e. dense nesting cover).
[2] Year of initial conversion to nesting cover.
[3] Category codes: n=native mix; t=tame mix; b=burned; h=hayed; s=spring burn; f=fall burn; 1-8=age since last management activity (YPM).
[4] – indicates projects were not sampled and includes projects with 1 YPM in 2001 and those older that 8 YPM.

Appendix B. Adjacent land-use categories and grouped designation.

Land Use Category	Group
Spring-seeded cereal crop	CROP
Fall-seeded cereal crop	CROP
Flax crop	CROP
Corn crop	CROP
Pea crop	CROP
Canola crop	CROP
Fallowed (bare soil)	CROP
Native area without livestock grazing (includes tree cover)	IDLE
Native area with livestock grazing (includes tree cover)	IDLE
Dense nesting cover	IDLE
Hay crop	IDLE
Farm infrastructure	IDLE
Recreational infrastructure	IDLE

Appendix C. *A priori* management and vegetation model suites and landscape covariates used for modelling small mammal and beetle abundance and biomass.

A Priori Management Models
Null (NULL)
Year (YR)
Treatment (TRT)
Year post management (YPM)
TRT+YPM
TRT+YPM+TRT*YPM
YR+TRT
YR+YPM
YR+TRT+YPM
YR+TRT+YPM+TRT*YPM

A Priori Vegetation Models
Null (NULL)
Visual obstruction reading (VOR)
Duff depth (DUF)
Maximum height (MXHT)
Forb composition (FORB)
Quadratic VOR (VOR+VOR^2)
Quadratic DUF (DUF+DUF^2)
Quadratic MXHT (MXHT+MXHT^2)
Quadratic FORB (FORB+FORB^2)
VOR+ DUF
VOR+ MXHT
VOR+ FORB
DUF+MXHT
DUF+ FORB
MXHT+FORB

Landscape Covariates
Null (NULL)
Conserved soil moisture (CSM)
Percent cropland (CROP)
Amount of planted cover (DNC)
Wetland area (WA)
Shrew CE (SHRW)
Deer mice CE (MICE)
Vole CE (VOLE)
Beetle CPUE (BEET)
Carabid CPUE (CARB)
Quadratic CSM (CSM+CSM^2)
Quadratic CROP (CROP+CROP^2)
Quadratic DNC (DNC+DNC^2)
Quadratic wetland area (WA+WA^2)
Quadratic shrew CE (SHRW+SHRW^2)
Quadratic deer mice CE (MICE+MICE^2)
Quadratic vole CE (VOLE+VOLE^2)
Quadratic beetle CPUE (BEET+BEET^2)
Quadratic Carabid CPUE (CARB+CARB^2)

Appendix D. Coleopteran families and species identified by Dr. G.E. Ball from voucher specimens collected between May and July in 2000 and 2001.

Family and Species

Carabidae (Ground Beetles)
- *Agonum cupreum*
- *Agonum cupripenne*
- *Amara cupreolata*
- *Amara obesa*
- *Amara patruelis*
- *Amara quenseli*
- *Amara torrida*
- *Anisodactylus harrisi*
- *Aphodius fimetarius*
- *Aphodius pinguis*
- *Blethisa multipunctata*
- *Brachinus cyanochroaticus*
- *Calosoma calidum*
- *Calosoma lepidum*
- *Calosoma moniliatum*
- *Carabus maeander*
- *Carabus serratus*
- *Carabus taedatus*
- *Chlaenius interruptus*
- *Chlaenius pupuricollis*
- *Chlaenius sericeus*
- *Dicaelus sculptilis upiodes*
- *Diplocheila striatopunctata*
- *Harpalus ventralis*
- *Patrobus lecontei*
- *Poecilus lucublandus*
- *Pterostichus adstrictus*
- *Pterostichus novus*

Cicindellidae (Tiger Beetles)
- *Cicindela limbalis*
- *Cicindela nebraskana*
- *Cicindela repanda*

Byrrhidae (Pill Beetles)
- *Byrrhus americanus*

Coccinellidae (Lady Bird Beetles)
- *Coccinella septempunctata*

Curculionidae (Snout Beetles)
- *Notaris bimaculatus*

Dytiscidae (Predacious Diving Beetles)
- *Agabus ontarionis*

Elateridae (Click Beetles)
- *Ctenicera aeripennis*
- *Ctenicera destructor*

Histeridae (Hister Beetles)
- *Hister interruptus*
- *Hister solaris*

Hydrophilidae (Water Scavenger Beetles)
- *Hydrochara obtusata*

Meliodae (Blister Beetles)
- *Lytta nuttalli*
- *Lytta viridiana*

Scarabidae (Dung Beetles)
- *Bolboceras sp.*
- *Eucanthus sp.*
- *Onthophagus hecate*
- *Onthophagus nuchicornis*
- *Phyllophaga sp.*
- *Serica intermixta*

Silphidae (Carrion Beetles)
- *Heterosilpha ramosa*
- *Nicrophorus hybridus*
- *Nicrophorus vespilloides*
- *Thanatophilus lapponicus*
- *Thanatophilus sagax*

Tenebrionidae (Darkling Beetles)
- *Eleodes hispilabris*
- *Eleodes opaca*
- *Eleodes tricarinata*

Appendix E. *A priori* management and vegetation model suites and landscape covariates used for modelling predator activity.

A Priori Management Models	*Landscape Covariates*
Null (NULL)	Null (NULL)
Treatment (TRT)	Percent cropland (CROP)
Year post-management (YPM)	Amount of planted cover (DNC)
TRT+YPM	Wetland area (WA)
TRT+YPM+TRT*YPM	Geographic location (GEO)
	Conserved soil moisture (CSM)
A Priori Vegetation Models	Vole CE (VOLE)
Null (NULL)	Beetle CPUE (BEET)
Visual obstruction reading (VOR)	Carabid CPUE (CARB)
Duff depth (DUF)	Carabid biomass (CBIO)
Maximum height (MXHT)	Quadratic CSM (CSM+CSM^2)
Percent forb cover (FORB)	Quadratic CROP (CROP+CROP^2)
Quadratic VOR (VOR+VOR^2)	Quadratic DNC (DNC+DNC^2)
Quadratic DUF (DUF+DUF^2)	Quadratic wetland area (WA+WA^2)
Quadratic MXHT (MXHT+MXHT^2)	Quadratic vole CE (VOLE+VOLE^2)
Quadratic FORB (FORB+FORB^2)	Quadratic beetle CPUE (BEET+BEET^2)
VOR+DUF	Quadratic Carabid CPUE (CARB+CARB^2)
VOR+MXHT	Quadratic Carabid biomass (CBIO+CBIO^2)
VOR+FORB	
DUF+MXHT	
DUF+FORB	
MXHT+FORB	

Appendix F. *A priori* management and vegetation model suites and landscape covariates used for modelling nest survival probabilities.

A Priori Management Models
Null (NULL)
Year (YR)
Treatment (TRT)
Year post management (YPM)
TRT+YPM
TRT+YPM+TRT*YPM
YR+TRT
YR+YPM
YR+TRT+YPM
YR+TRT+YPM+TRT*YPM

A Priori Vegetation Models
Null (NULL)
Visual obstruction reading (VOR)
Duff depth (DUF)
Maximum height (MXHT)
Forb composition (FORB)
Quadratic VOR (VOR+VOR^2)
Quadratic DUF (DUF+DUF^2)
Quadratic MXHT (MXHT+MXHT^2)
Quadratic FORB (FORB+FORB^2)
VOR+ DUF
VOR+ MXHT
VOR+ FORB
DUF+MXHT
DUF+ FORB
MXHT+FORB

Landscape Models
Null (NULL)
Conserved Soil Moisture (CSM)
Percent cropland (CROP)
Amount of planted cover (DNC)
Wetland area (WA)
Beetle CPUE (BEET)
Vole CE (VOLE)
Quadratic CSM (CSM+CSM^2)
Quadratic CROP (CROP+CROP^2)
Quadratic DNC (DNC+DNC^2)
Quadratic wetland area (WA+WA^2)
Quadratic beetle CPUE (BEET+BEET^2)
Quadratic vole CE (VOLE+VOLE^2)
CSM+VOLE
CROP+VOLE
DNC+VOLE
WA+VOLE
BEET+VOLE
CSM+CSM^2+VOLE
CROP+CROP^2+VOLE
DNC+DNC^2+VOLE
WA+WA^2+VOLE
BEET+BEET^2+VOLE
CSM+VOLE+VOLE^2
CROP+VOLE+VOLE^2
DNC+VOLE+VOLE^2
WA+VOLE+VOLE^2
BEET+VOLE+VOLE^2
CSM+CSM^2+VOLE+VOLE^2
CROP+CROP^2+VOLE+VOLE^2
DNC+DNC^2+VOLE+VOLE^2
WA+WA^2+VOLE+VOLE^2
BEET+BEET^2+VOLE+VOLE^2